A COARSE SCREENING PROCESS FOR EVALUATION OF THE EFFECTS OF LAND MANAGEMENT ACTIVITIES ON SALMON SPAWNING AND REARING HABITAT IN ESA CONSULTATIONS

Technical Report 94-4

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A COARSE SCREENING PROCESS FOR POTENTIAL APPLICATION IN ESA CONSULTATIONS

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"Scholars concerned about ecological integrity face a serious dilemma. Because ecological knowledge is incomplete, prediction of the consequences of societal action is an inexact science. With a desire to appear objective and reasonable, ecologists accept the proposition that 'tight causal proof is the only basis for instituting controls' that will protect ecological health (Woodwell, 1989). Woodwell concludes, and I agree, that a hesitancy to accept a preponderance of evidence (as opposed to causal proof) and a willingness to compromise stringent requirements for avoiding biotic impoverishment is the 'epitome of unreasonableness.' He continues: 'Indulgence in the hyperobjectivity now being pushed on science lends support to avarice [while it] destroys the credibility of science and scientists as a source of common sense'..."

Karr, J.R. 1992. Ecological integrity: protecting earth's life support systems. p. 223-238. In: R. Costanza, B.G. Norton, and B.D. Haskell (eds.). 1992. Ecosystem Health. New Goals for Environmental Management.

"If there is to be any chance of abating the loss of biodiversity, action must be taken immediately...The indispensable strategy for saving our fellow living creatures and ourselves in the long run, is, as the evidence compellingly shows, to reduce the scale of human activities...Unless humanity can move determinedly in that direction, all of the efforts now going into in situ conservation will eventually lead to nowhere, and our descendants' future will be at risk."

Ehrlich, P.R. and Wilson, E.O., 1991. Biodiversity Studies: Science and Policy, Science, **253**: 758-762.

ABSTRACT

Spring, summer, and fall chinook salmon in the Snake River Basin have been listed as "endangered" under the Endangered Species Act (ESA) by the National Marine Fisheries Service (NMFS). The ESA requires that activities authorized, funded, or carried out by federal agencies do not adversely modify critical habitat for listed species. The interim policy of NMFS is that the aggregate effect of all land use activities should result in improved habitat conditions and survival for the listed salmon species. The Coarse Screening Process provides objective, measurable criteria to evaluate the consistency of single and combined land management activities with these legal and policy goals. Although salmon populations are affected by a variety of activities throughout the migratory range of the listed salmon, the Coarse Screening Process focuses only on land management activities and their effect on salmon survival in spawning and rearing habitat.

The Coarse Screening Process relies on three sets of criteria. Biologically-based habitat standards are used to determine the need for improvement in habitat conditions. Land management standards are used to determine the consistency of activities with protection and improvement of habitat conditions and, in some cases, are contingent on habitat conditions. The screening process also requires that data exist for all land management and habitat conditions set as standards that can potentially be affected by single or combined activities. Under the screening process, activities are deemed consistent with ESA habitat policies only when all three sets of criteria are satisfied.

Potential habitat standards were reviewed for their effects on salmon survival and production, their linkages to management activities, and their relevance to conditions in the Snake River Basin. Where land management standards could adequately protect key habitat attributes, they were set in lieu of quantitative habitat standards. Habitat standards were recommended based on the habitat requirements of the listed salmon. Habitat attributes reviewed for their potential utility as screening elements included metrics for channel substrate, pools, large woody debris, bank stability, water temperature, miscellaneous pollutants, water quantity and timing. Quantitative habitat standards were recommended for channel substrate, water temperature, and bank stability. It is recommended that where these standards are not met, that any activity that can potentially delay improvement in habitat condition should be deferred or curtailed until the habitat standard is met or a statistically improving trend is documented through monitoring over at least five years.

Approaches to developing habitat standards based on the "range of natural variability" were reviewed, but are not recommended because such approaches do not adequately protect salmon populations.

Potential land management standards were reviewed for their effects on salmon habitat. Land management standards were recommended for riparian reserves, estimated sediment delivery, roads, grazing, and roadless reserves. Approaches based on "Equivalent Clearcut Areas" were not recommended as land management standards.

Application of the screening process to the John Day, Umatilla, and Clearwater River Basins is recommended to provide refugia for potential salmon colonists to and from the area currently

designated as critical habitat.

Land management standards are recommended to remain in effect until habitat conditions in at least 90% of the managed watersheds in the Snake River Basin either meet habitat standards or exhibit a statistically significant improving trend as documented through monitoring over at least 5 years.

The screening process should be applied at scales representing logical units of salmon production, which may generally include watersheds of approximately 4th to 6th order streams.

ACKNOWLEDGEMENTS

Many helped us in this task. They deserve none of the blame for any of deficiencies, excesses, or errors in this document, but they do deserve credit for their help. The following people helped us by supplying data, literature, important discussions, and/or helpful reviews: Dr. Leslie Bach (Umatilla National Forest), Dale Bambrick (Yakama Indian Nation), David Bayles (Pacific Rivers Council), Dr. Robert Beschta (Oregon State Univ.), Dr. Lee Benda (Univ. of Washington), Paul Boehne (Wallowa-Whitman National Forest), Dr. Daniel Bottom (Oregon Department of Fish and Wildlife), Kelly Burnett (USFS PNW Research Station), Dr. Don Chapman (Don Chapman Consulting Inc.), Travis Coley (USFWS), Robert Gill (Wallowa-Whitman National Forest), Dr. David Burns (Payette National Forest), Dr. Christopher Frissell (Univ. of Montana), Charles Huntington (Clearwater Biostudies), Jeffrey Lockwood (NMFS), Richard Jones (Clearwater National Forest), Dr. James Karr (Univ. of Wash.), John Kelley (Confederated Tribes and Bands of the Warm Springs Indian Reservation of Oregon), Bruce McIntosh (USFS PNW Research Station), Dr. Philip Mundy (Consulting Fishery Biologist), Dr. Charles Petrosky (Idaho Department of Fish and Game), and Michael Purser (Confederated Tribes of the Umatilla Indian Reservation) and the helpful staff of the Columbia River Inter-Tribal Fish Commission: Doug Dompier, Keith Hatch, Matthew Schwartzberg, Roberta Stone, and Jim Weber.

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PREFACE

This report assumes that the reader has a working knowledge of the diverse fields that are broached in putting the synthesized information together. Unfortunately, discipline-specific jargon, that may not be familiar to some readers, is used throughout the manuscript. Although we regret inconveniences that this may pose, we realized that avoidance of technical jargon or inclusion of definitions would further bloat a large report. We trust that the trail of bread crumbs that can be found in the bibliography will provide ample guidance that the ambitious reader can use to find greater detail and definitions for jargon.

The citations included in the text are not necessarily the definitive citations for a given point. Rather citations are made on the basis that they amply support a given point. While efforts were made to comprehensively review and synthesize available literature, an exhaustive review and discussion of available information and literature was far beyond the scope of this report. Books can and have been written on specific topics (e.g. statistical detection or channel metamorphosis) covered in the report; we trust that diligent readers will turn to the references where more detail is desired.

Habitat data used in the report were chosen on the basis of completeness and availability. As with citations, we did not attempt to collect, compile, and analyze all available, and often fragmentary, habitat data in the Columbia River Basin.

In an effort of this magnitude, errors undoubtedly remain. We apologize for them in advance. They are ours, alone.

Finally, it is patently obvious that we diligently tried not to allow notions of what might be "politically acceptable" to factor into our evaluations and recommendations, although at times, we gave it momentary consideration. In our collective experience, passing off politically-tempered recommendations that may be inadequate to solve problems as "scientifically sound" is one of the most serious problems existing in natural resource management. It has damaged natural resource science, natural resource management, and society at large, but most of all, it has damaged the resources. Politically and socially mediated management decisions have, are, and should be made, but they should not be passed off as "scientific" when they are not.

OVERVIEW

The Endangered Species Act (ESA) requires that activities authorized, funded, or carried out by federal agencies do not adversely modify critical habitat for listed species. The interim policy direction of NMFS under the ESA is that the aggregate effect of all land use activities should result in improved habitat conditions that increase the survival for the listed salmon species in the Snake River Basin. We have taken these policy goals at face value in designing a screening process aimed at both avoiding incremental habitat damage and requiring that adequate approaches are undertaken to improve damaged habitat.

The screening process employs three sets of filters to assess the consistency of land management activities with improvement and protection of habitat conditions: biologically-based habitat standards, land use standards, and data availability (Table A). These filters represent decision criteria. Activities should be considered to be consistent with the protection and improvement of salmon habitat only when they comply with all aspects of the three sets of filters.

The biologically-based habitat standards (Table B) are used as a filter to assess habitat conditions and the need for changes in land management activities so that they are consistent with the goal of improving degraded habitat conditions and salmon survival. The biologically-based habitat standards are a core set of measurable habitat variables, chosen based on the linkages among salmon survival, habitat conditions, and land use activities. Where existing conditions do not comply with the biologically-based habitat standards, it is likely that salmon survival has been reduced by a combination of natural- and management-induced conditions. In these cases, the screening process requires passive restoration. Passive restoration is defined as curtailing and deferring activities that contribute to or forestall the recovery of poor habitat conditions. Active restoration, such as road obliteration, should also be considered as a means to speed habitat recovery, in watersheds that do not meet habitat standards.

Land use standards (Table B) were developed to avoid the degradation of salmon habitat and/or the foreclosure of future management options in the effort to protect and restore salmon habitats and populations. These land use standards can be used to screen out land management practices that generally have negative effects on salmon habitat over time. These land use standards were developed based on the review and synthesis of available information on the ecological functions of watersheds, the effects of land disturbance, the downstream response of habitat conditions to these effects, and salmon response to habitat alteration.

The screening process also provides a framework for establishing minimum monitoring requirements for habitat evaluation. If insufficient data exist, activities should be deferred or curtailed until data on conditions set as standards are collected and summarized.

Although the screening process is based on a synthesis of available information, there is still considerable uncertainty that combined measures will result in habitat improvement over the short term, given existing watershed conditions and continued landscape-level disturbance. The most risk-free and effective approach to habitat improvement is to fully protect all watersheds from continued land disturbance and undertake only low-risk restoration activities, such as road obliteration, until monitoring indicates that substantial improvement has occurred in the majority of damaged watersheds. Deviation from this approach increases the risks of further habitat damage and failing to achieve habitat improvement. The costs of failing to protect and improve habitat may be incalculable because additional habitat degradation or the maintenance of poor habitat conditions will contribute to the extirpation of geographically isolated salmon populations. We strongly recommend that a low risk approach be taken to allowing habitat to recover. However, the screening process does not provide the lowest risk approach. Instead, we have taken a fairly mechanistic approach to protecting salmon habitat. Mechanistic approaches have some potential for failure because they are always based on incomplete information about complex and chaotic systems. In order to address issues of risk and uncertainty, the screening process implicitly and explicitly puts most of the burden of proof on demonstrating that habitat conditions affecting survival have improved.

The screening process relies on "adaptive management" through monitoring. If habitat conditions improve or comply with biologically-based habitat standards, land disturbing activities can go forward, provided they comply with land use standards (Table A). If habitat conditions deteriorate, passive and active restoration efforts should be re-doubled. We recommend that land use standards remain in place until there is a strong and geographically robust signal documented through monitoring that habitat conditions in most managed watersheds in the Snake River Basin have improved or meet the biologically-based habitat standards.

We recommend that the screening process be applied to watersheds outside of the area currently designated as critical habitat for the listed salmon species to provide refugia for salmon colonists to and from critical habitat. There is a high likelihood of rapid extirpation of many endemic populations due to the combined effect of existing conditions in various habitats occupied by salmon during their lifecycle. Potential sources of colonists need to be protected so that barren habitats can be recolonized. Areas outside of critical habitat can provide potential refugia for salmon colonists from and to critical habitat. Therefore, we recommend that the screening process be applied to the John Day, Clearwater, and Umatilla River Basins.

1.0 BIOLOGICALLY-BASED HABITAT STANDARDS

While many different pieces are essential to a healthy salmon habitat, we have focused on those elements that have been shown to have the greatest influence on salmon survival in their natal habitat. The habitat elements set as standards do <u>not</u> represent a comprehensive list of parameters that are useful for monitoring the effects of land management on habitat conditions and survival; they are a minimum set of habitat variables to be used to determine the need to alter land management practices consistent with providing habitat conditions conducive to salmon survival. Habitat elements were set as numeric standards only if all of the following criteria were met: a) research has consistently shown that the habitat variable strongly influences salmon survival and production; b) data indicate that the habitat variable has affected salmon survival and production in Snake River Basin watersheds; c) the preponderance of data indicates a linkage between land use activities and condition of the habitat variable; d) there is no viable land use standard that can be set to adequately assure that the condition of the habitat variable will be protected or improved; and e) some measurable change in the variable can expected over time in response to changes in land management. Where land use standards could adequately protect or restore vital habitat attributes, they were adopted as coarse screening elements in lieu of habitat standards to reduce monitoring and analysis, and expedite screening. Again, we stress that exclusion of parameters from the core set of habitat standards does not necessarily reflect on its utility for monitoring purposes. Other parameters can and should be monitored.

The biologically-based habitat standards were not stratified by land or channel types because available information does not indicate that the *response of salmon to specific habitat conditions* significantly varies regionally by land or channel types. Stratification issues are salient to questions of attainability and channel response to perturbation, but are not necessary to identify habitat conditions needed by salmon to survive. Meeting the habitat requirements of salmon species must be the biological bottom line of efforts to protect and restore spawning and rearing habitat consistent with efforts to stabilize and restore the listed salmon runs.

The recommended habitat standards may not be attainable in all systems, even under completely natural conditions. In systems where recommended habitat standards are not attainable, it is critical that existing conditions not be exacerbated by land use activities, because it may contribute to the extirpation of salmon species from marginal and sensitive habitats. In some sensitive watersheds, such as those in the highly erosive Idaho batholith, tolerable salmon habitat conditions and significant land disturbance may be mutually exclusive.

Natural disturbances and processes, such as fire and floods, can contribute to departures from the habitat standards that may temporarily reduce salmon survival. Because such departures can affect salmon survival, it is critical that efforts be taken to reduce the anthropogenic contributions that increase the magnitude or duration of the departures from standards.

Alternatives to a biologically-based approach have been proposed for the development of standards for salmon habitat variables. The utility of developing habitat standards based on ranges of natural variation is discussed and evaluated in Section 1.5.

1.1 CHANNEL SUBSTRATE: FINE SEDIMENT AND COBBLE EMBEDDEDNESS

Salmon survival and production are reduced as fine sediment increases, producing multiple negative impacts on salmon at several lifestages. Increased fine sediment entombs incubating salmon in redds, reduces egg survival by reducing oxygen flow, alters the food web, reduces pool volumes for adult and juvenile salmon, and reduces the availability of rearing space for juveniles rendering them more susceptible to predation. Reduced survival-to-emergence (STE) for the listed salmon caused by elevated fine sediment increases is of particular concern because it is a source of density-independent mortality that can have extremely significant negative effects on salmon populations even at low seeding.

The rearing capacity of salmon habitat is decreased as cobble embeddedness levels increase. Overwinter rearing habitat may be a major limiting factor to salmon production and survival in parts of the Snake River Basin. The loss of overwintering habitat may result in increased levels of mortality during rearing lifestages.

Data indicate that elevated levels of fine sediment are a problem throughout Snake River Basin habitats. The recovery of degraded substrate conditions is unlikely unless estimated sediment delivery is reduced to less than 20% over natural.

Recommendation: Although a wide variety of metrics have been used to measure and express fine sediment levels, we recommend setting a fine sediment standard for surface fines, due to the effects surface sediment size has on the sedimentation of redds, and relative ease and rapidity of monitoring. However, fine sediment conditions at depth also reduce STE. Sole reliance on a standard for surface fine sediment may **not** adequately address fine sediment conditions at depth. Therefore, fine sediment at depth should be monitored to determine the trends in substrate conditions affecting salmon survival, even though we do not recommend it as a mandatory standard for screening. While a surface fine sediment standard may not be completely adequate for documenting all trends affecting salmon survival, it should be adequate for screening land use activities. Reducing surface fine sediment should contribute to improvement in substrate conditions at depth, over time, but this cannot be assumed.

Available data from the Snake River Basin indicate that STE is substantially reduced for chinook salmon at levels greater than 20% surface fine sediment. However, any increase in fine sediment levels in spawning habitat probably represents a deleterious modification of critical habitat.

We recommend that watersheds should be managed so that surface fine sediment levels average less than 20% in spawning habitat and no increase occurs where surface fine sediment averages less than 20% in spawning habitat.

We recommend that watersheds should be managed so that cobble embeddedness averages less than 30% within rearing habitat and no increases occur where cobble embeddedness averages less than 30% in rearing habitat.

It is extremely unlikely that channel substrate conditions can be protected or recovered without adequate control of sediment delivery at the watershed scale. Therefore, we also recommend standards for sediment delivery as part of substrate standards. Additional detail on sediment delivery can be found in Section 3.2.

We recommend the following standards for watersheds where substrate standards are not met and sediment delivery is estimated to be more than 20% over natural: 1) Reduce sediment delivery through suspension of on-going activities and prohibition of the initiation of activities that can increase erosion over natural levels (See Table 1), and implement active restoration measures (e.g., road obliteration) as needed, until substrate conditions meet standards or a statistically significant ($p<0.05$) improving trend over at least 5 years is documented through monitoring <u>and</u> total sediment delivery from anthropogenic sources is estimated to be less than 20% over natural. 2) If substrate conditions do not meet standards after total sediment delivery is estimated to be less than 20% over natural and substrate conditions have exhibited a statistically significant, improving trend over at least five years, activities that can increase erosion should be implemented/re-initiated only when combined with active and passive restoration measures such that they result in net reductions in sediment delivery until substrate conditions meet standards.

We recommend the following for watersheds where substrate standards are not met but total sediment delivery from all anthropogenic sources is estimated to be less than 20% over natural: 1) Eliminate on-going activities and prohibit activities that can increase erosion over natural levels and implement active restoration measures, such as road obliteration, until substrate conditions meet standards or a statistically significant ($p<0.05$) improving trend over at least 5 years is documented through monitoring. 2) Once substrate conditions have exhibited a statistically significant trend over at least five years, activities that increase erosion should be implemented/re-initiated only when combined with active and passive restoration measures such that they result in net reductions in sediment delivery, until substrate conditions meet standards.

We recommend the following standards for watersheds where substrate standards are met but sediment delivery is estimated to be more than 20% over natural: Activities that increase erosion over natural levels should be implemented or continued only when combined with active and passive restoration measures that result in net reductions in sediment delivery, until sediment delivery is less than 20% or substrate conditions exhibit a statistically significant ($p<0.05$) improving trend over at least five years as documented by monitoring.

If substrate conditions show statistically significant deterioration ($p<0.40$) over any period longer than one year, activities that increase erosion should be suspended until substrate conditions return to their initial condition prior to deterioration; active restoration measures aimed at reducing sediment loads should also be undertaken in such cases. The low level of statistical significance ($p<0.40$) regarding habitat deterioration is recommended as part of a risk adverse approach and because of the high environmental and economic costs associated with failing to address habitat degradation once it has occurred. (For more detail, see Section 1.6 Notes on Statistical Significance).

1.2 CHANNEL MORPHOLOGY

1.2.1 Pools and Large Woody Debris (LWD)

Available data indicate that the production of salmon is reduced as pool frequency and volume decrease. Large pools are required by salmon during rearing, spawning, and migration. Pools provide thermal refugia, velocity refugia during storm events, resting habitat for migrating salmon, and important rearing habitat for juvenile salmon.

Fine sediment is deposited in pools during waning flows. Residual pool volume is the volume of a pool not filled by fine sediment accumulations. Fine sediment volumes in pools reduce pool quality and reduce residual pool volumes (the pool volume available for salmon use).

Available data indicate that salmon production increases as LWD increases. LWD provides cover, velocity refugia, and plays a vital role in pool formation and the maintenance of channel complexity required by salmon in natal habitat. LWD also aids in reducing channel erosion and buffering sediment inputs by providing sediment storage in headwater streams. Existing data indicate that extensive pool loss in salmon habitat over the past 50 years has occurred and is a major problem in the managed watersheds of the Snake River Basin. Existing information amply indicates that LWD levels in streams have been reduced by land management activities. This reduction in LWD has probably contributed to the significant pool losses documented in some Snake River Basin streams.

Recommendation: Although monitoring pool volume and frequency is an important habitat assessment tool, we do not recommend a numeric standard for pool frequencies. The most efficient and ecologically effective way to manage for pools is to limit and/or reduce sediment delivery and provide full protection of LWD sources, riparian vegetation, floodplains, and bank stability. These measures should be undertaken regardless of current pool frequency status. Trends in pools are highly unlikely to be temporally sensitive indicators of habitat trends. It is highly probable that pool formation and loss proceed slowly except during more extreme, and relatively rare, events such as flooding and/or major landsliding. It is probable that, over time, adequate pool volumes and frequencies will result with full protection of the areas within riparian reserves and by meeting standards for channel substrate, residual pool volumes, bank stability, and sediment delivery, although it cannot be assumed that this will be the case. If trends indicate decreasing pool frequencies and volumes, extra efforts should address the likely causes of pool loss: lack of riparian vegetation and attendant bank instability, elevated sediment delivery, and the loss of LWD. We do recommend that pool volumes and frequencies be monitored for trends, although we do not recommend a numeric standard.

We recommend against the construction of pools in channel segments. These efforts are unlikely to have much longevity because they fail to address the causes of pool loss.

We recommend that watersheds should be managed so that there is a decrease in fine sediment volumes in pools and increased residual pool volumes in managed watersheds. We recommend monitoring these pool variables because they hold considerable promise as "early warning indicators" of trends in pool frequency and volume. We do not recommend a numeric standard for residual pool volumes because linkages between residual pool volumes and fish production have not been well established. Where there is an increasing trend in fine sediment volumes in pools, sediment delivery should be reduced through passive and/or active watershed restoration.

Rather than specify numeric standards for in-channel LWD, we recommend full protection of LWD recruitment systems through the establishment of riparian reserves as a surrogate approach. The recommended riparian reserves should protect existing LWD sources and allow re-establishment of LWD recruitment over time. Trends in LWD will probably not be rapidly established and are not amenable to a timely monitoring/management feedback approach. Activities should be screened for their effect on LWD recruitment over time, not on the basis of current LWD levels.

In some watersheds with highly degraded riparian zones, active watershed restoration approaches aimed at re-establishing tree stocking will be necessary to restore LWD recruitment. Although we do not recommend setting LWD levels as a standard in the screening process, we do recommend that LWD be monitored for trends.

LWD additions have limited effectiveness and benefits to salmonids unless the causes of poor habitat conditions are adequately addressed; they also have potential negative effects in degraded streams. We recommend that LWD additions aimed at forming pools should only be undertaken where: 1) it is ecologically appropriate given the types and existing conditions of the streams and riparian vegetation; 2) the causes of pool loss and channel instability have been adequately addressed, e.g., loss of LWD sources and bank stability, etc; and 3) it has been documented that all other habitat conditions are amenable to salmon survival and production (e.g., water temperature). LWD additions should never be considered a surrogate for the protection of riparian vegetation or the control of sediment delivery.

Elevated sediment delivery can lead to the loss of usable pool volume and decreased pool frequency. We recommend that sediment delivery should be reduced to less than 20% over natural in order to protect against pool loss from sedimentation.

1.2.2 Bank stability

Bank stability is of prime importance in maintaining habitat conditions favoring salmon survival. Bank instability increases channel erosion that can lead to increased levels of fine sediment and the in-filling of pools. Unstable banks can lead to stream incisement that can reduce baseflow contributions from groundwater and increase water temperature. Bank instability can cause channel widening that can significantly exacerbate seasonal water temperature extremes and destabilize LWD.

Recommendation: We recommend that watersheds should be managed so that more than 90% of channel banks on all streams in a watershed are stable; bank stability should be maintained where it is greater than 90%. In watersheds where bank stability is less than 90%, or there is a decreasing trend in bank stability, activities that can potentially decrease bank stability or forestall recovery should be eliminated until the standard has been reached or a statistically significant ($p<0.05$) improving trend over at least five years has been documented through monitoring. Once an improving trend has been established but the standard is not met, activities should only be allowed if they do not impede continued improvement in bank stability. Suspension of riparian grazing is one of the key strategies to restoring bank stability.

We recommend against mechanical channel stabilization methods. These approaches can shift bank instability problems downstream and tend to fix channels to positions within floodplains, thwarting the ability of a stream to create complex habitat features, such as side channels and meander bend pools. Mechanical bank stabilization approaches are ecologically unsound and can create more and worse problems than they are aimed at solving.

1.3 WATER QUALITY

1.3.1 Water Temperature

Available information indicates that the elevation of summer water temperatures impairs salmon production at scales ranging from the reach to the stream network and puts fish at greater risk through a variety of effects that operate at scales ranging from the individual organism to the aquatic community level. Maximum summer water temperatures in excess of 60°F impair salmon production. However, many smaller streams naturally have much lower temperatures and these conditions are critical to maintaining downstream water temperatures. At the stream system level, elevated water temperatures reduce the area of usable habitat during the summer and can render the most potentially productive and structurally complex habitats unusable. Decreases in winter water temperatures also put salmon at additional risk. The loss of vegetative shading is the predominant cause of anthropogenically elevated summer water temperature. Channel widening and reduced baseflows exacerbate seasonal water temperature extremes. Elevated summer water temperatures also reduce the diversity of coldwater fish assemblages.

Elevated summer water temperature is a significant problem throughout many of the streams in the Snake River Basin. Decreased winter water temperature may also be a problem in salmon habitat in some parts of the Snake River Basin.

Recommendation: We recommend that watersheds should be managed to increase the downstream extent of summer water temperatures that are suitable to salmon. Where daily maximum summer water temperatures in excess of 60°F exist in historically usable spawning and rearing habitat for salmon, passive restoration measures should be taken to reduce water temperatures and active restoration should be undertaken, where likely to be effective in speeding the natural recovery of water temperatures. Activities that have the potential to increase water temperatures or forestall the recovery of natural water temperatures should not be allowed on any stream. The establishment of the recommended riparian reserves (Sec. 3.1) should be effective in protecting against additional water temperature increases from vegetation loss. Active restoration will be necessary in some heavily damaged riparian zones if adequate water temperature regimes are to be restored. Restoration of water temperature regimes should focus on the restoration and protection of riparian vegetation and hydrologic regimes.

The protection of riparian reserves is recommended in lieu of a stream shading standard. Activities that decrease or forestall the recovery of shading should not be allowed. In streams draining managed watersheds, an increasing trend in shading should occur. Although we do not recommend stream shading as a standard for screening purposes, we do recommend that stream shading be monitored for trends.

1.3.2 Miscellaneous Pollutants

A wide variety of water quality conditions affect salmon, although a full review is beyond the scope of the screening process. Existing state and federal water quality standards may not be sufficient to protect and restore salmon populations.

Recommendation: We recommend that state and federal water quality standards should be reviewed and revised to fully protect and restore salmon. In the interim, although existing water quality standards may be inadequate, they should be enforced. Watersheds should be managed to meet all applicable state and federal water quality standards.

We recommend that transport of toxic materials along spawning and rearing reaches should be restricted or eliminated because history indicates that the spill of transported toxic material can cause high levels of salmon mortality. Storage of toxic materials upstream of spawning and rearing areas should be eliminated due to high risks associated with spills.

1.4 WATER QUANTITY AND TIMING

The frequency and magnitude of stream discharge strongly influence substrate and channel morphology conditions, as well as the amount of available spawning and rearing area for salmon. Increased peak flows can cause redd scouring, channel widening, stream incisement, increased sedimentation. Lower streamflows are more susceptible to seasonal temperature extremes in both winter and summer. The dewatering of reaches can block salmon passage. It also appears that flows exert a strong control on smolt survival through the mainstem to the ocean. Cumulatively, streamflows in spawning and rearing habitat affect seasonal water availability and flows in the mainstem Columbia River.

Groundwater is a seasonally important source of streamflow in most alluviated, valley-fill systems. Groundwater pumping can decrease baseflow.

Recommendation: Although we cannot currently recommend numeric standards for seasonal flows, flow decreases during the low flow period can have several negative effects on natal salmon habitat and may also constrain mainstem options. Therefore, we recommend that no additional withdrawals of surface water or groundwater should occur in any streams tributary to waterways that provide salmon spawning, rearing, or migration habitat, until studies have ascertained that current flows are more than adequate for salmon survival and the maintenance of favorable habitat conditions. We recommend that studies should be undertaken to determine the instream flow levels needed for salmon in each watershed, including assessment of the streamflows needed to maintain channel morphology, sediment routing, floodplain function, and adequate water temperatures, as well as the streamflows needed for salmon passage, rearing, and spawning. The studies should also analyze the current, cumulative effects of existing upstream water withdrawals on downstream flow availability on the Columbia River mainstem and the effects on salmon survival. Where instream flows are inadequate for salmon or for the restoration and maintenance of favorable habitat conditions, efforts should be made to acquire water for instream salmon and habitat needs.

We also recommend that all wetlands be fully protected from adverse soil and vegetation impacts. Efforts should be made to restore baseflow regimes by restoring the function of degraded meadow systems. Implementation of our recommendations on constraining vegetation removal at the watershed scale based on sediment delivery, together with establishment of riparian reserves, protection of roadless areas, and suspension and alteration of riparian grazing, and reductions in road mileage should provide some protection against increased peakflows and decreased baseflows.

1.5 THE USE OF "RANGES OF NATURAL VARIABILITY" AS AN ALTERNATIVE APPROACH TO THE DEVELOPMENT OF HABITAT STANDARDS

Though some work has suggested managing streams within the "range of natural variability," such an approach has limited promise for the protection of fragile resources such as weak populations of salmon that are geographically isolated. A "range of natural variability approach" (RNVA) does not address the habitat conditions needed to allow salmon recovery. Instead, a RNVA deems habitat conditions acceptable as long as they fall within statistically determined ranges, even though the conditions may be neither natural nor conducive to salmon survival. Due to its reliance on statistical detection of significant change in highly variable phenomena, a RNVA inherently allows degraded conditions to persist until either the effect is pronounced and persistent or a large sampling effort is complete. For many resources, statistically significant degradation may not be detected until damage is severe and/or irreversible; in such cases, sole reliance on statistical methods of detection prior to changing management is bound to fail to protect fragile resources. The listed salmon and their habitats are just such cases.

There are a number of formidable barriers to applying RNVAs. Data on the frequency, duration, and extent of habitat conditions are currently lacking. It may not be possible to collect applicable data, because very few large, low gradient streams have habitat conditions that reflect solely natural conditions. Notably, larger order, low-gradient streams are typically the most potentially productive and structurally diverse habitats. The same habitats are the most sensitive to **cumulative** degradation due to their reach characteristics and position within watersheds. Most watersheds that are in natural condition are smaller, steeper, higher elevation systems; stream conditions and processes in these watersheds were probably never representative of natural conditions found in larger, lower elevation stream systems. Therefore, the potential utility of a RNVA is limited to smaller streams. Limitations on potential data sources are compounded because robust watershed classification schemes that can be used to extrapolate habitat condition frequency distributions have not been tested. Although the RNVA purports to address the attainability of habitat conditions, existing barriers to implementation/extrapolation render that claim dubious. The only reliable manner of ascertaining the natural variability and attainability of habitat conditions in a given watershed is to have the complete watershed in a natural state and monitor habitat conditions over time.

Recommendation: Although a RNVA may ultimately have some promise as a habitat management tool, it is clear that it is not currently an operational tool because of existing barriers and problems. Due to its reliance on statistical detection and failure to address the habitat requirements of salmon, it has limited potential for aiding in the recovery of salmon populations and puts salmon at serious risk. Therefore, we do not recommend using an RNVA.

1.6 NOTES ON STATISTICAL SIGNIFICANCE

Recommendation: We recommend a fairly high level of statistical significance for detection of improving trends ($p<0.05$) so that there is a fairly high probability that habitat conditions have actually improved prior to initiating activities that can potentially reduce the rate of habitat recovery. We also recommend a fairly low level of statistical significance for the detection of deteriorating trends in habitat variables set as standards ($p<0.40$) in order to elicit prompt management responses to deteriorating habitat conditions. This relatively low level of statistical significance decreases the magnitude of the minimum detectable effect (MDE) at a given sample size, level of variability, and statistical power. The differing levels of significance are based on consideration of the ecological costs associated with the different types of errors involved in accepting potentially false hypotheses. The most rational approach to setting levels of statistical significance and power is to base them on the expected costs associated with management decisions based on the errors of accepting different false hypotheses as true. It is anticipated that the costs of allowing poor habitat conditions to be maintained or further deteriorate far outweigh the costs of continuing to take a risk adverse approach to habitat protection and improvement even after it may have improved because the maintenance of poor habitat conditions is likely to have irreversible effects on isolated spawning populations of salmon and the effects of land disturbing activities are slowly reversible.

The magnitude of the MDE is a function of variability, sample size, and statistical power, as well as the specified level of statistical significance. There are statistical compromises involved in trying to adjust each of these factors. We recommend that the factors be adjusted through sample design or the selection of power so that MDE for accepting that deterioration has occurred ($p<0.40$) is no greater than a 10% deterioration in the initial value of the variable. Some variables, such as water temperature, can have much smaller MDEs at the recommended level of statistical significance; in the case of testing for deterioration, monitoring efforts should be designed to make the MDEs as small as possible.

2.0 THE ROLE OF PASSIVE AND ACTIVE RESTORATION IN IMPROVING HABITAT CONDITIONS

Recommendation: Complete passive restoration should occur in all watersheds that do not meet the recommended habitat standards. In some damaged watersheds, active restoration will also be necessary if habitat conditions are to be improved. Active restoration is unlikely to improve habitat in the absence of passive restoration. Active restoration measures should be sustainable, facilitate ecosystem function, focus on the causes of degradation, and reconnect linkages between aquatic, riparian, and upland functions.

3.0 PERFORMANCE STANDARDS FOR LAND MANAGEMENT

Reliance on habitat standards, alone, is not adequate to protect salmon populations and habitat from damage over time. Under such an approach, causes of habitat damage are only addressed after habitat degradation has occurred and has been documented. The avoidance and prevention of habitat damage is more effective financially, operationally, and biologically than attempting to reverse degradation. Protection of habitat from degradation is critical because habitats cannot recover unless the root causes of degradation are arrested. The recovery times from degradation are extremely long and may require centuries, even once the causes of habitat degradation are eliminated. Weak salmon populations may never recover from reduced survival caused by additional habitat degradation. Therefore, land management standards that have some promise for avoiding degradation are more effective than relying solely on *ex post facto* feedback from monitoring.

The performance standards for land management activities have been developed based on existing information on ecosystem function, and the effects of land use on habitat conditions and salmon survival. Activities that do not comply with the land use standards increase the risk of habitat damage, forestall the recovery of damaged habitats, and/or foreclose future options for habitat protection and improvement. We recommend using the land use standards as screening elements to determine the consistency of activities with the goals of protecting and restoring habitats for the listed salmon.

Although a great deal is known about the many linkages, some uncertainty exists regarding the exact response of salmon populations and habitat to the cumulative effects of land management over time. The recommended land use standards are based on common sense principles that <u>should</u> govern decision-making under uncertainty: favoring actions that are robust to uncertainty; monitoring results; and favoring actions that are reversible and do not foreclose management options. Existing information and case histories indicate that protection measures and habitat improvement are rapidly reversible, while underprotection and further degradation may not be. Therefore, some of the land use standards are based on the standard engineering practice of using "factors of safety" as a hedge against uncertainty over time.

We recommend that all land use standards should remain in place until the listed salmon populations have recovered or the habitat conditions in >90% of managed watersheds with either meets standards or has improved. This requirement is aimed at providing some geographical assurance that a well-distributed population of relatively high quality habitats is in place to provide refugia for salmon colonists to and from the Snake River Basin. This geographic improvement criterion is also aimed at requiring that there is a geographically robust data signal indicating that combined activities have been effective in reversing degradation prior to considering the initiation of activities that may cause additional degradation or foreclose future management options, such as entry into roadless areas or altering riparian reserves.

3.1 RIPARIAN RESERVES

Riparian areas provide functions indispensable to the maintenance of high quality habitat conditions. Undisturbed riparian vegetation provides sediment detention, stream shading, LWD sources, and bank stability. Riparian vegetation also regulates air temperatures in the stream environment. Disturbance and vegetation removal within riparian zones reduce stream shading, increase sediment delivery, reduce bank stability, compact soils, and reduce LWD; these changes exacerbate seasonal temperature extremes, cause channel widening, increase fine sediment, and reduce channel complexity. Vegetation removal in riparian zones may decrease the longevity of riparian forest stands by increasing the frequency and magnitude of blowdown.

Headwater streams comprise the bulk of the channel network in watersheds with salmon habitat; they play a pivotal role in shaping downstream habitat conditions. These small streams are the most susceptible to the adverse effects of vegetation removal.

Floodplains are temporal extensions of the stream channel. Streams migrate across floodplains over time. During floods, floodplains act as channels. Therefore, floodplains should receive the same protection as streams.

Aside from the inherent functions provided by riparian zones, riparian reserves are needed to ameliorate the effects of disturbance within watersheds. Many riparian systems are degraded and are not fully functional.

<u>Recommendation</u>: We recommend that riparian reserves along all streams extend at least 300 feet in slope distance from the outer edges of the floodplains (or stream edge in the absence of floodplains), or to the topographic divide, whichever is less. Reserves of this width on all streams, should maintain and restore stream shading, bank stability, LWD levels over time, and insulate streams from air temperature alterations, <u>provided the riparian reserves are fully functional</u>. Riparian reserves of 300 feet may not fully protect riparian vegetation from increased windthrow nor adequately buffer streams and floodplains from high levels of sediment delivery from upslope activities or extreme events. Recommended reserve widths may also not protect against water temperature increases and changes in baseflow hydrology because activities outside of the reserves, such as roadcuts, can disrupt subsurface hydrology.

Many riparian areas have been degraded. This will impair the effectiveness of the riparian reserves in protecting and restoring habitat conditions. It is unlikely that expanded reserves can offset the continuing effects of impacts within the reserves. Therefore, we recommend that active restoration efforts focus on eliminating the persistent effects from existing impacts, such as roads, within the reserves.

We recommend that no additional anthropogenic disturbance should occur within the riparian reserves. Grazing should be suspended within the reserves in watersheds where the temperature standards are not met, until the standard is met, or a statistically significant improving trend ($p<0.05$) over at least five years is documented through monitoring. Grazing should be suspended within half a tree height from the edge of floodplains (or streams when floodplains are absent), in all reaches or watersheds where bank stability standards are not met, until the standard is met or a statistically significant improving trend ($p<0.05$) over at least five years is documented through monitoring. Temporary elimination of riparian grazing in degraded reaches and watersheds is probably the most effective approach to restoring riparian systems and realizing rapid habitat improvement in the Snake River Basin. Where all habitat standards are met, riparian grazing should be tightly controlled and closely monitored.

We do not recommend riparian restoration approaches involving the removal of vegetation. Such approaches are fraught with risk, have low reversibility, and their effectiveness remains a matter of speculation. Such approaches should not be considered until they have been documented to have been successful under ecologically applicable experimental conditions, or that habitat and riparian conditions have improved in the majority of Snake Basin watersheds that provide salmon habitat. Riparian restoration efforts should focus on activities that are low risk and likely to be effective, such as suspension of grazing in degraded reaches.

3.2 SEDIMENT DELIVERY

Information generally indicates that elevated sediment delivery leads to increases in fine sediment and cobble embeddedness in salmon habitat, although the exact functional response varies due to a wide variety of factors. Maintenance of elevated sediment delivery can prevent recovery in streams with degraded substrates. Although limited, data indicate that substrate conditions are unlikely to improve when sediment delivery is estimated to be greater than about 20% over natural. Available information indicates that sediment loads in excess of 20% over natural lead to increased levels of fine sediment and cobble embeddedness or maintain poor substrate conditions.

Sediment delivery is a critical landscape issue to address because it is a dominant control on stream channel substrate conditions that can reduce salmon survival, even at low seeding levels. Elevated sediment delivery also leads to pool volume loss and channel widening. Channel widening exacerbates seasonal temperature extremes, even without the loss of vegetation. Elevated sediment delivery has degraded salmon habitat in many watersheds in the Snake River Basin.

Recommendation: The recommended standards for sediment delivery can be found in Section 1.1 Channel Substrate.

Until sediment delivery from grazing is incorporated into existing models, recommended sediment delivery levels (i.e., reducing sediment delivery to less than 20% over natural) are based on the assumption that grazing will be suspended in degraded watersheds. In watersheds where grazing continues to occur, efforts should be made via field sampling and modeling to incorporate sediment delivery from grazing into estimates of total watershed sediment delivery.

The use of modeled estimates of sediment delivery as a management tool is fraught with the potential for error and abuse. However, available models have been widely used in the Snake River Basin and, occasionally, validated. Despite the drawbacks inherent in a modeling-based approach, it appears to be the most viable tool for constraining land disturbance to levels that allow recovery and prevent degradation via sedimentation. Due to its widespread usage, we recommend that sediment delivery be initially estimated by models based on the R1-R4 Sediment Yield model adapted so that the following sources of sediment delivery are included: 1) mass failure and surface erosion from logging; 2) mass failure and surface erosion from all existing roads, **regardless of age**; 3) channel and surface erosion from grazed lands; and 4) surface and mass erosion from mining.

In-channel habitat conditions are the bottom line. Where substrate conditions deteriorate or degraded conditions persist, efforts to reduce sediment delivery should continue until substrate sediment conditions meet the standards or exhibit a statistically significant, improving trend ($p<0.05$) over at least 5 years as documented through monitoring.

3.3 LOGGING-RELATED DISTURBANCE: EQUIVALENT CLEARCUT AREA (ECA) APPROACHES

Due to the low reversibility of land use effects both on-site and off-site, it is desirable to limit land disturbance to levels that adequately protect habitat from degradation. Information and data indicate that habitat degradation generally increases as the amount of disturbance within a watershed increases. However, at a given level of land disturbance, other factors strongly influence the magnitude, duration and extent of habitat damage that occurs.

ECA approaches do not adequately represent many factors that influence the amount of degradation caused by disturbance, such as proximity to streams and geomorphic sensitivity. ECA approaches mask the amount of riparian disturbance. ECA approaches assume recovery times that do not reflect actual ecological recovery times in riparian zones. ECA approaches ignore important potential sources of cumulative effects, such as grazing and mining. In grazed areas, significant habitat damage occurs even at extremely low ECA levels. These failures of ECA approaches are significant because most managed watersheds have had considerable impacts in riparian zones and grazing is almost ubiquitous in the Snake River Basin. In many watersheds, poor habitat conditions indicate that cumulative impacts have been significant even at ECA levels that have been deemed to confer a low risk of cumulative effects to watersheds.

Recommendation: We do not recommend using ECA approaches to limit land disturbance, due to their multiple failures. Instead, we recommend limiting the magnitude of land disturbance spatially via riparian and roadless reserves, and, in magnitude, via other screening standards, such as sediment delivery, although these approaches also have drawbacks.

3.4 ROADS

Roads are one of the greatest sources of habitat degradation. Roads significantly elevate on-site erosion and sediment delivery, disrupt subsurface flows essential to the maintenance of baseflows, and can contribute to increased peakflows. Roads within riparian zones reduce shading and disrupt LWD sources for the life of the road. These effects degrade habitat by increasing fine sediment levels, reducing pool volumes, increasing channel width and exacerbating seasonal temperature extremes.

Recommendation: We recommend that existing road mileage in all watersheds should be reduced consistently, over time, especially within riparian reserves. No additional roads should be constructed, until, at least, monitoring has indicated that the habitat conditions in at least 90% of watersheds in the Snake River Basin have improved or meet standards.

We recommend that roads that will not be re-located or obliterated should be improved to reduce sediment delivery and improve drainage, except where the hazards of improvement pose short-term threats to salmon that may not be outweighed by the long-term benefits of road improvement. In such situations, roads should be closed/obliterated/relocated and culverts removed.

3.5 GRAZING

Livestock grazing can widen channels, reduce shade, elevate sedimentation, and exacerbate seasonal water temperature extremes. Livestock grazing has caused significant habitat degradation in the Snake River Basin. Suspension of riparian area grazing is the grazing strategy that is most compatible with re-vegetation and salmon habitat recovery. There is a low probability that any grazing management system will allow recovery to begin in damaged riparian systems without some period of rest. Most widely-used grazing practices are incompatible with the protection and restoration of aquatic ecosystems. Grazing retards recovery in degraded riparian systems. Exclusion of livestock from riparian zones has been shown to increase summer baseflow. Forage utilization standards are an ineffective approach to restoration and protection in degraded reaches, wet meadows, seeps, and travel corridors because habitat damage stems from trampling and chiseling of banks and vegetation by livestock as well as the browsing and grazing of vegetation. A more effective approach to habitat improvement is to eliminate grazing in these areas.

Recommendation: As with other types of land disturbance that cause increased erosion, we recommend that grazing be temporarily suspended in watersheds that do not meet substrate standards until the standards are met, or a statistically significant ($p<0.05$) improving trend over the course of 5 years is documented through monitoring and total sediment delivery is estimated to be less than 20% over natural. Livestock grazing in watersheds where water temperature standards are not met in salmon habitat should be suspended within the riparian reserves until water temperature standards are met or a statistically significant ($p<0.05$) improving trend over at least 5 years is documented through monitoring. In watersheds where bank stability standards are not met, we recommend that grazing be temporarily suspended within half a tree height of floodplains (or streams when floodplains are absent) until the bank stability standard is met, or monitoring documents that a statistically significant ($p<0.05$) improving trend has occurred over at least five years. In many areas, riparian area grazing is difficult to control; in these areas it will be necessary to completely remove livestock from watersheds to prevent grazing within floodplains and reserves until recovery occurs or standards are met.

We recommend that livestock be restricted from access to spawning reaches during and after the spawning season, because livestock can trample redds when they ford streams. If livestock access to these reaches cannot be prevented during the spawning and incubation periods, they should be removed from watersheds prior to the onset of the spawning season.

We also recommend that grazing be eliminated from environments where it is clearly incompatible with the protection of aquatic resources. Grazing in wet meadows with fine-grained, non-cohesive soils and without woody bank vegetation almost always leads to stream damage. Therefore, we recommend that such environments not be subjected to grazing unless fenced and all habitat standards are met.

Where grazing continues or is re-initiated after degraded conditions have improved, use should be tightly controlled and closely monitored. Monitoring is required in affected riparian areas that are grazed and in downstream habitat affected by upstream grazing.

Although lowered forage utilization rates do have some utility in reducing the impacts of livestock on aquatic habitats, we do not recommend relying solely on lowered forage utilization rates to provide adequate levels of habitat protection. The control of forage utilization, alone, does not adequately address bank trampling, soil compaction, sedimentation and restoration of riparian plant assemblages and status.

3.6 ROADLESS RESERVES

Available information indicates that much of the Snake River Basin has been degraded. Existing roadless and wilderness areas provide the only high-quality habitats and islands of natural functioning systems left in the Snake River Basin. The extent of these areas is limited. It may not be possible to enter roadless systems without compromising their natural function and/or without degrading habitat conditions. Roadless, unlogged tracts form the cornerstones of habitat recovery efforts. Continued diminishment of areas functioning somewhat naturally increases the risk of failing to improve habitat conditions at scales ranging from the reach to the region.

Recommendation: Given existing habitat degradation and uncertainties, it is prudent to require that most of the degraded habitat be improved prior to taking risks with the scarce areas having high quality habitat. We recommend that roadless tracts greater than 1000 acres should not entered, at least, until monitoring documents that habitat conditions in >90% of managed watersheds either meet habitat standards or have exhibited statistically significant improvement over at least five years. Smaller roadless tracts may also have important ecological value. We recommend that smaller roadless tracts should not be disturbed unless it can be shown through peer-reviewed analysis that the disturbance will not affect habitat conditions, impede habitat recovery, or foreclose options for habitat recovery.

3.7 GEOGRAPHIC CRITERIA FOR RE-EVALUATING LAND USE STANDARDS

We recommend using "adaptive management" concepts in the application of the screening process. That is, management approaches should be adjusted as more knowledge is gained and conditions change. Given this approach, the standards for land use set in the screening process should be subject to revision as conditions change. Specifically, less restrictive land use standards may be considered after habitat conditions have improved considerably. However, premature relaxation or over-relaxation of protection measures has a risk of reversing or preempting habitat improvement. Careful thought should be given to defining a reasonable point at which higher risk approaches may be considered. Land use standards should only be revised once efforts to improve degraded habitats have been successful over a wide range of geography and conditions.

Recommendation: We recommend that the land use standards should not be revised to allow activities with higher risk or that foreclose management options until habitat conditions in at least 90% of the managed watersheds in the Snake River Basin either meet all biologically-based habitat standards, or have shown a statistically significant improving trend over at least a five period as documented by monitoring.

There are several reasons behind the recommendation. First, it is only prudent to ensure that habitats and survival have improved significantly prior to foreclosing management options via entering roadless areas or altering vegetation within riparian reserves because of the biologically perilous condition of the listed salmon species, the existing widespread degradation of habitats, and uncertainty regarding improvement in habitat and survival. Once 90% of the managed watersheds have been documented to have improved, additional risks are, at least, slightly more tenable. Second, it is prudent to ensure that information indicating habitat improvement is data driven and geographically robust. Third, higher risk approaches should not be considered until improved habitats are well-distributed throughout the basin in order to provide refugia and protection for groups of potential salmon colonists. Fourth, there should be some indication that there is greater system resiliency in habitat conditions throughout the basin. Our recommendation provides some assurance that many habitats have greater resilience via watershed recovery and that there is better system resilience in case some habitats are degraded by the re-initiation of high risk activities or natural catastrophes. For these reasons, we recommend that land use standards should not be relaxed until at least 90% of the managed habitats have improved as described above.

4.0 SCREENING AT THE WATERSHED SCALE OR FOR INDIVIDUAL ACTIVITIES

The coarse screening process can be used at the watershed scale (See Table A) to determine the consistency of cumulative activities with the goals of protecting and improving salmon habitat and survival. The process should be applied at watershed scales that represent logical units of salmon production and encompass both rearing and spawning habitat. Generally, experience indicates that the watersheds of streams that range from 3rd to 6th order at the watershed outlet can serve as logical production/management units. In some situations it may be desirable to apply the coarse screening process at smaller and larger watershed scales due to the salmon's use of the habitats or environmental conditions within a watershed. Thus, the actual scale of application will have to be dictated by field conditions. In many cases, the choice of watershed scale for analysis of cumulative effects on salmon habitat has already been made. In some cases, these previously selected scales should suffice for application of the screening process. We recommend that the core set of land management standards be applied as screening elements at all scales, on streams of all orders, to determine the consistency of activities with the goal of protecting and improving salmon habitat and survival. The coarse screening process can also be used to evaluate the consistency of a single activity with the goals of protecting and improving salmon habitat and survival. However, watershed scale data for habitat conditions set as standards that can be affected by the activity are still required to apply the process for a single activity.

When screening at the watershed scale, data must be available and integrated on a watershed basis for all habitat variables set as standards. If data do not exist, all on-going activities that can potentially affect the parameters should be temporarily suspended or deferred until habitat data are collected and summarized. The potential effects of activities can be found in Table 1.

Data are required for land use parameters set as standards. If data do not exist, all activities that disturb vegetation and soils should be temporarily suspended or deferred until data are collected on land use status. Data requirements related to land use standards include the status and condition of roadless reserves, riparian reserves, roads, sediment delivery, and how current land use activities relate to the land management standards, e.g., whether on-going activities are within the riparian reserves.

The spatial distribution of habitat and watershed conditions is a key information need. A geographic information system data base is recommended for presenting data at the watershed scale, although it is not required as part of the screening process.

Where data indicate that habitat standards are not met, activities that can prevent or forestall habitat recovery should be considered to be inconsistent with improvement of salmon habitat improvement and should be suspended or deferred. Table 1 provides an overview of activities and their potential effects on habitat conditions. Complete passive restoration at the watershed scale should proceed until trend data indicate habitat recovery. Where likely to be effective, active restoration measures should be undertaken to improve habitat conditions.

Where data indicate that an activity does not comply with standards for land management, it should be considered inconsistent with habitat protection and improvement and be modified to be consistent with land management standards, curtailed, or deferred. Active and passive restoration should be taken to move watershed conditions towards compliance with standards, unless trend data indicate that habitat standards are met.

Table A. Flowchart for the Coarse Screening Process

Data Availability

In-channel habitat[1] data are available for variables set as standards.

No → Defer or curtail all activities that can potentially affect attainment of standards (See Table 1) until data are collected and summarized.

Yes ↓

Watershed condition/land management data[2] are available.

No → Defer or curtail all activities that can potentially affect attainment of standards until data are collected and summarized.

Yes ↓

Standards Compliance Within Watershed

In-channel habitat conditions meet standards or a statistically significant improving trend exists.

No → Defer or curtail all activities that can forestall recovery. Continue to implement all land management standards[3]. Undertake active restoration activities as needed. Monitor in-channel habitat parameters to establish baseline conditions[4].

Yes ↓

Land management meets or exceeds the land management standards set for screening.

No → Defer or curtail activities that do not comply with land management standards. Modify activities consistent with land management standards.

Yes ↓

Standards Compliance Within Region

≧ 90% of all managed watersheds within the spawning and rearing habitat for the listed species have a documented, statistically significant improving trend in all in-channel habitat standards or meet all standards (i.e., in-channel habitat and land management standards).

No → Continue to implement recommended land management standards and take active restoration measures as necessary to improve habitat conditions.

Yes ↓

Consider whether land management standards can be relaxed, yet still be consistent with maintenance of habitat conditions conducive to excellent salmon survival.

Proceed with consultation.

Endnotes

1. Recommended in-channel habitat standards include surface fine sediment, cobble embeddedness, bank stability, water temperature. All these standards should be measured within salmon-bearing habitat of the watershed and calculated on a weighted area basis for screening purposes at a logical salmon production scale (e.g., 4-6th order watersheds). These standards should also be measured in lower order subwatersheds as an early warning device to prevent propagation of effects downstream, to document recovery, and to validate sediment and temperature models, although this is not a requirement in screening (see companion Monitoring document).

2. Watershed or land management standards are recommended for sediment delivery, riparian reserves, grazing, roads, riparian grazing, and roadless reserves.

3. Land management standards suggested in csp should be implemented regardless what the current habitat condition is. These are standards that have high likelihood of maintaining existing high quality conditions where they occur and serve as a foundation for allowing recovery to proceed.

4. Baseline conditions for in-channel habitat should be established for fine sediment at egg pocket depth, large woody debris, pool frequency and volume, residual pool volume, stream shading. For these habitat conditions, standards are not proposed. However, these variables have great biological significance and degradation of these variables or lack of improvement from the current baseline in the context of general watershed rehabilitation for managed watersheds may constitute cause for concern and increased effort at restoration.

TABLE B. COARSE SCREENING PROCESS SUMMARY OF STANDARDS

In-Channel Habitat Conditions

Element and Standard	Condition	Related Land Management Standards
Channel Substrate		
Surface fine sediment (diam.<0.25 in.): Average surface fine sediment #20% in spawning habitat; no increase where already #20%. Cobble embeddedness (CE): Average CE #30% in rearing habitat; no increase where already #30%.	Substrate standards met and estimated sediment delivery #20% over natural.	No increase in sediment delivery from single or combined activities. All new activities with potential to produce sediment offset by at least equivalent sediment abatement via active restoration.
	Substrate standards met but estimated sediment delivery >20% over natural	Reduce sediment delivery via passive and active restoration[1] to #20% over natural. All new activities that increase erosion should be combined with sediment abatement measures that result in a net reduction in sediment delivery.
	Substrate standard exceeded and estimated sediment delivery >20% over natural; fine sediment or CE increase.	Reduce sediment delivery via complete passive restoration until sediment delivery #20% over natural and substrate conditions meet standards or exhibit a statistically significant (p<0.05) improving trend over $5-yr period. Implement active restoration as needed to reduce sediment delivery and improve substrate conditions.
	Substrate standard exceeded and estimated sediment delivery #20% over natural.	Reduce sediment delivery via complete passive restoration until substrate conditions meet standards or exhibit a statistically significant (p<0.05) improving trend over $5-yr period. Implement active restoration as needed to reduce sediment delivery and improve substrate conditions.
Fines by depth: Although not set as a numeric standard, monitoring substrate trends at depth is recommended as part of adaptive management.	Regardless of condition.	Monitor fines by depth in key areas to evaluate correlation with surface fines by area.

Coarse Screening Process Summary of Standards (Continued)

In-Channel Habitat Conditions

Element and Standard	Condition	Related Land Management Standards
Channel Morphology		
Large Woody Debris (LWD): Land management standards set in lieu of numeric standard.	Regardless of condition.	Fully protect vegetation and soils within riparian reserves, meet bank stability standards, and Monitor LWD. Implement active restoration as needed to reestablish natural LWD recruitment from riparian zones. Add LWD to streams only after causes of LWD loss have been adequately addressed and where ecologically appropriate.
Pool frequency and volume: Land management standards set in lieu of numeric standard.	Regardless of condition.	Fully protect vegetation and soils within riparian reserves, meet bank stability standards and limit/reduce sediment delivery to #20% over natural. Monitor pool frequency and volume. Add LWD only after causes of pool loss have been adequately addressed and where ecologically appropriate.
Residual pool volume: Not set as a numeric standard. Achieve an increasing trend in residual pool volume.	Regardless of condition.	Fully protect vegetation and soils in riparian reserves, meet bank stability standards, and limit/reduce sediment delivery to #20% over natural. Monitor residual pool volume.
	Declining trend from baseline.	Reduce sediment delivery via passive and/or active restoration.
Bank stability: Bank stability on all streams average $90%; no decrease in bank stability when >90%.	Average bank stability <90% or a decrease in bank stability.	Implement passive restoration (suspension of grazing within half a tree height of floodplains or streams) until bank stability meets standard or exhibits a statistically significant (p<0.05) improving trend over $5 years. Implement active restoration addressing causes of bank instability as needed to improve bank stability. Do not mechanically stabilize banks (e.g., rip-rap).
	Average bank stability $90%.	Apply appropriate management controls to maintain bank stability.

Coarse Screening Process Summary of Standards (Continued)

In-Channel Habitat Conditions

Element and Standard	Condition	Related Land Management Standards
Water Quality		
Water temperature: Maximum daily water temperature #60°F in historically usable spawning and rearing habitat.	Maximum daily water temperature >60°F.	Fully protect vegetation in riparian reserves. Implement complete passive riparian restoration to reduce water temperature. Suspend grazing within riparian reserves until water temperature meets standard or exhibits a statistically significant improving trend over $5 years. Implement active restoration as needed to improve water temperature conditions.
	Maximum daily water temperature #60°F.	Fully protect vegetation within riparian reserves from any additional impacts. Control and monitor on-going activities within riparian reserves to assure they do not increase water temperatures.
Stream shading: Land management standards set in lieu of a numeric standard.	Regardless of condition.	Fully protect vegetation within riparian reserves. Activities that decrease shading or forestall recovery should not be allowed on any stream.
Misc. Pollutants: Review and revise current state and federal water quality standards as needed to adequately protect salmon. In the interim, meet current water quality standards.	Regardless of condition.	Monitor water quality parameters set as state and federal water quality standards where the potential for pollution exists. Eliminate or restrict transport of toxic materials along and upstream of spawning and rearing reaches. Eliminate storage of toxic materials within watersheds with spawning or rearing habitat.
Water Quantity and Timing: Not set as numeric standards.	Regardless of condition.	Suspend additional groundwater and surface water withdrawals in watersheds with spawning and rearing habitat until studies determine instream flows are more than adequate for salmon production and survival, and maintenance and restoration of favorable habitat conditions. Where flows are inadequate, acquire instream flows. Protect and restore wetlands and degraded meadow systems. Fully protect vegetation and soils within riparian and roadless reserves.

Coarse Screening Process Summary of Standards (Continued)

Land Management Standards

Element and Standard	Condition	Related Land Management Standards
Sediment delivery: Sediment delivery #20% over natural.	(Additional details under **Channel Substrate**)	
Riparian reserves: $300 ft slope distance from floodplains, or stream where floodplains do not exist, or to topographic divide, whichever is less.	Regardless of condition.	Fully protect vegetation and soils within riparian reserves from any additional anthropogenic disturbance. Do not implement approaches to riparian restoration involving vegetation removal until they have been shown to be effective under applicable ecological conditions. Reduce existing road mileage in reserves. Improve road drainage within reserves. Active restoration should focus on reducing impacts within riparian reserves where needed to improve habitat conditions.
	Water temperature standard exceeded.	Suspend grazing within riparian reserves until water temperatures meet standard or exhibit a statistically significant improving trend over at least 5 years.
	Bank stability standard not met.	Suspend grazing within half a tree height from the edge of floodplains, or streams when floodplains are absent, until bank stability meets standard or exhibits a statistically significant improving trend over at least 5 years.
	All habitat standards met.	Carefully control and monitor all on-going activities within riparian reserves to assure degradation does not occur.
Equivalent Clearcut Area (ECA): Not recommended as a land use standard for limiting land disturbance.		Limit land disturbance via riparian and roadless reserves, application of in-channel habitat standards, and sediment delivery standard.

Coarse Screening Process Summary of Standards (Continued)

Land Management Standards

Element and Standard	Condition	Related Land Management Standards
Roads: Decrease road mileage in managed watersheds. Improve drainage and decrease sediment delivery from roads that will not be obliterated or re-located.	Habitat conditions in <90% of managed watersheds either meet standards or have exhibited statistically significant improvement.	Defer construction of new roads. Continue to upgrade, obliterate, or re-locate existing roads.
	Habitat conditions in >90% either meet all habitat standards or have exhibited statistically significant improvement.	Consider re-evaluation of prohibition on new road construction.
Grazing: Forage utilization standards not recommended as a numeric standard.	Regardless of condition	Eliminate livestock access to spawning reaches during spawning and incubation periods.
	Substrate standards exceeded.	Suspend grazing within watershed until sediment delivery is reduced via passive and active restoration to #20% over natural and substrate conditions meet standards or exhibit a statistically significant improving trend over $5-yr period.
	Bank stability standard not met.	Suspend grazing within half a tree height from the edge of floodplains, or streams when floodplains are absent, until bank stability meets standard or exhibits a statistically significant improving trend over at least 5 years.
	Water temperature standard exceeded.	Suspend grazing within riparian reserves until water temperatures meet standard or exhibit a statistically significant improving trend over at least 5 years.
	All habitat standards met.	Carefully control and monitor all grazing within riparian reserves to assure degradation does not occur.

Coarse Screening Process Summary of Standards (Continued)

Land Management Standards

Element and Standard	Condition	Related Land Management Standards
Roadless reserves: Maintain all roadless tracts >1000 acres undisturbed; maintain smaller roadless tracts undisturbed until documented that disturbance will not forestall habitat recovery nor foreclose management options.	Habitat conditions in <90% of managed watersheds either meet all habitat standards or have exhibited statistically significant improvement.	Maintain protection of roadless tracts.
	Habitat conditions in >90% of managed watersheds either meet all habitat standards or have exhibited statistically significant improvement.	Consider re-evaluation of roadless reserves.
Spatial Criteria for Re-evaluation of Land Management Standards: Habitat conditions in >90% of managed watersheds either meet all habitat standards or have exhibited statistically significant improvement.	Habitat conditions in <90% of watersheds meet all habitat standards or have exhibited a statistically significant improving trend in all habitat elements over $5-yr period.	Maintain integrity of riparian and roadless reserves and continue to implement screening process. Monitor trends in habitat condition for the entire set of watersheds within the Snake River Basin for trends. Implement active restoration as needed to improve habitat conditions.
	Habitat conditions in >90% of watersheds meet all habitat standards or have exhibited a statistically significant improving trend in all habitat elements over $5-yr period.	Consider re-evaluation of land use standards (e.g., riparian and roadless reserves).

Endnotes for Summary

1. Passive restoration is defined as curtailing or deferring activities that contribute to degraded conditions or forestall natural recovery processes. <u>Complete</u> passive restoration is defined as curtailing and deferring all activities at the watershed scale that contribute to degradation or potentially forestall recovery. In the case of channel substrate, complete passive restoration entails curtailing and deferring all ground disturbing activities that increase erosion over natural levels.

Active restoration is defined as taking actions to reduce or eliminate the effects of existing impacts, such as obliterating or upgrading roads.

INTRODUCTION

A coarse screening process is proposed as a means to assess the consistency of land management activities with the goal of improving salmon habitat conditions and resultant survival of salmon species listed under the Endangered Species Act (ESA). The ESA requires that activities authorized, funded, or carried out by federal agencies do not adversely modify critical habitat for listed species. The interim policy direction of National Marine Fisheries Service (NMFS) under the ESA is that the aggregate effect of all land use activities should result in improved habitat conditions that increase the survival of the listed salmon species in the Snake River Basin. We have taken these policy goals at face value in designing a screening process aimed at both avoiding incremental habitat damage and requiring that approaches are undertaken to improve damaged habitat. The screening process is designed to lead to habitat improvement and protection by screening out activities that have a risk of adversely affecting habitat conditions, especially when those conditions are negatively affecting salmon survival.

The screening process employs three basic sets of filters to assess the consistency of land management activities with improvement and protection of habitat conditions: <u>biologically-based habitat standards</u>, <u>land use standards</u>, and <u>data availability</u>. These filters represent decision criteria. In order to protect habitat from adverse modification, activities should be undertaken or continued only when they comply with all aspects of the three basic filters. That is, actions should go forward only when it has been documented that habitat conditions are not averse to salmon survival and that activities within the watershed will neither forestall the recovery of degraded conditions nor degrade conditions vital to salmon survival. Incremental improvement in individual activities, relative to the recent past, may not result in any improvement in habitat conditions and salmon survival because: a) many management activities have been extremely poor; b) cumulative effects continue to accrue in fish habitat due to extremely poor watershed conditions; and c) many habitat conditions are far beyond points where "less worse" will allow recovery. For these reasons, biologically-based habitat standards are used in the screening process to ensure that habitat conditions vital to salmon survival are improved, rather than relying solely on incremental improvements in land management activities as a means to improve habitat.

The biologically-based habitat standards are used as a filter to assess habitat conditions and the need for changes in land management activities so that they are consistent with the goal of improving degraded habitat conditions and salmon survival. The biologically-based habitat standards are a core set of measurable habitat variables. They were chosen based on their established links to salmon survival in spawning and rearing habitat and the regional robustness of the linkages among salmon survival, habitat conditions, and land use activities. In cases where application of land use standards could adequately protect or restore vital habitat attributes, they were adopted as coarse screening elements in lieu of habitat standards in order to reduce monitoring and analysis, and expedite screening at the watershed scale. Where existing conditions do not comply with the biologically-based habitat standards, it is likely that salmon survival has been reduced by a combination of natural and management induced conditions. In these cases, the screening process invokes a low risk approach until habitat improvement has occurred: activities that can forestall habitat recovery should be deferred or curtailed and active restoration undertaken, until habitat conditions improve or habitat standards are met. "Passive restoration" is defined as curtailing and deferring activities that can forestall natural habitat recovery.

Land use standards were developed to avoid the degradation of salmon habitat and/or the foreclosure of future management options in the effort to protect and restore salmon habitats and populations. These land use standards serve as criteria to screen out land management practices that can have negative effects on salmon habitat over time. The standards were developed based on a synthesis of available information on the ecological functions of watersheds, the on-site impacts of land disturbance, the downstream response of habitat conditions to the effects of on-site impacts, and salmon response to habitat alteration. Knowledge of the general effects of activities on salmon habitat is enough to determine the consistency of the activities with the goal of improving salmon habitat and survival; determination of the exact functional response of habitat and survival to types of activities is not needed given the general goal of improvement in habitat and survival.

While a great deal is known about the many linkages among land uses and habitat conditions, there is some uncertainty regarding the exact response of salmon populations and habitat to the cumulative effects of land management over time. Therefore, the recommended land use standards and attendant screening process are based on important common sense principles that <u>should</u> govern decision-making under uncertainty: favoring actions that are robust to uncertainty over time; monitoring of results; and favoring actions that are reversible and do not foreclose management options (Ludwig et al., 1993). The body of existing information and case histories clearly indicate that protection measures and habitat improvement are rapidly reversible. In contrast, underprotection and further degradation may not be.

The screening process also provides a framework for establishing minimum monitoring requirements for habitat evaluation. If existing data are insufficient to determine watershed and habitat conditions and their relationship to standards, activities should be deferred or curtailed until data necessary to determine the condition of habitat variables set as standards are collected.

The screening process relies strongly on "adaptive management" through monitoring. If habitat conditions improve or comply with biologically-based habitat standards, land disturbing activities can go forward, provided they comply with land use standards. If habitat conditions deteriorate, passive and active restoration efforts should be re-doubled until habitat conditions meet the biologically-based habitat standards or have been documented to improve in the vast majority of managed watersheds. Although there will undoubtedly be pressure to relax land use standards, especially once <u>some</u> habitats have improved, we recommend that land use standards remain in place until there is a strong and geographically robust signal that the screening process, in combination with watershed restoration measures, has been effective in improving habitat conditions as documented through monitoring. Once monitoring indicates that the majority of watersheds in the Snake River Basin have recovered, land use standards can be re-evaluated.

We recommend that the screening process be applied to watersheds that are outside of, but adjacent to, the area currently designated as critical habitat for the listed species so as to provide refugia for salmon colonists to and from critical habitat. Because there is a high likelihood of rapid extirpation of many endemic populations due to the combined effect of existing conditions in various habitats occupied by salmon during their lifecycle, there is a need to protect potential sources of colonists so that barren habitats can be recolonized. Application of the screening process to areas outside of critical habitat can also aid in protecting potential refugia for salmon colonists from critical habitat. For these reasons, we recommend that the screening process be applied to the John Day, Clearwater, and Umatilla River Basins.

Although the screening process is based on a synthesis of available information, studies, and experiments, there is still considerable uncertainty that combined measures will result in habitat improvement over the short term, given existing watershed conditions and continued landscape-level disturbance. The most risk-free and effective approach to habitat improvement is to fully protect all watersheds from continued land disturbance and undertake only low-risk restoration activities, such as road obliteration, until monitoring indicates that substantial improvement has occurred consistently in the majority of damaged watersheds. Deviation from this conservative approach increases the risks of further habitat damage and failing to achieve habitat improvement. The costs of failing to protect habitat may be incalculable because additional habitat degradation or the maintenance of poor habitat conditions will contribute to the extirpation of numerous, spatially dispersed, and weak salmon populations, thereby reducing the effectiveness of remaining populations in supporting the species. We strongly recommend that a low risk approach be taken by restricting continued habitat impacts and allowing habitat to recover. However, the screening process does not provide the lowest risk approach. Instead, the screening process is based on a fairly mechanistic approach to protecting salmon habitat. We have emphasized those activities of high magnitude impact (a combination of activity intensity and spatial extent plus proximity to the species' instream habitat). Mechanistic approaches have some potential for failure because they are always based on incomplete information about complex and chaotic systems operating interactively at many spatial and temporal scales. In order to address issues of risk and uncertainty, the screening process implicitly and explicitly puts the burden of proof on demonstrating that habitat conditions affecting survival have improved.

The protection and improvement of conditions in spawning and rearing habitat can contribute to the increased survival of the salmon species currently listed under the ESA. However, it is not likely that it will be sufficient to arrest the continuing declines in the populations even if all habitats are restored to their highest sustainable potential, because passage mortality at downstream hydroelectric facilities remains high. Unless passage mortality is substantially and rapidly decreased, protection and improvement of natal habitat can only slow the rate of decline in salmon populations. However, it is also clear that the existing degradation of natal habitat has accelerated the decline of salmon populations in the Snake River Basin. While all sources of anthropogenically elevated salmon mortality must be reduced if salmon declines are to be arrested, the screening process is constrained to addressing conditions in spawning and rearing habitat.

1.0 BIOLOGICALLY-BASED HABITAT STANDARDS

While many different pieces are essential to a healthy salmon habitat, we have focused on those elements that have shown the greatest influence on salmon survival in their natal habitat. However, salmon cannot thrive without **all the pieces in place**. The lack of any one essential condition can eliminate salmon from a reach or a river. For instance, abundant pools, large wood, and excellent substrate conditions cannot completely offset the deleterious effects of exceedingly hot and/or polluted water. Land management should be focused on getting all the pieces in place by allowing watersheds to recover through the elimination and/or reduction of ecosystem impacts caused by land management.

Biologically-based habitat standards are recommended as one part of a coarse screening process. The habitat elements set as standards do not represent a comprehensive list of parameters that are useful for monitoring the effects of land management on habitat conditions and survival; rather, they are a minimum set of habitat variables to be used to determine the need to alter land management practices consistent with providing habitat conditions conducive to salmon survival. Many other terrestrial and aquatic variables besides those recommended as standards could and should be monitored to provide information on the status and trend of fish habitat and to elucidate linkages among land uses, climate, terrestrial processes, and aquatic conditions. However, the recommended habitat standards provide a core set of elements for identifying the need to improve habitat conditions and to evaluate the consistency of activities with habitat improvement goals in light of existing habitat conditions.

In evaluating their potential as biologically-based habitat standards, habitat elements were set as numeric standards only if all of the following criteria were met: a) research has consistently shown that the habitat variable strongly influences salmon survival and production; b) data indicate that the habitat variable has affected salmon survival and production in Snake River Basin watersheds; c) the preponderance of data indicates a linkage between land use activities and condition of the habitat variable; d) there is no viable land use standard that can be set to adequately assure that the condition of the habitat variable will be protected or improved; and e) some measurable change in the variable can expected over time in response to changes in land management.

The set of habitat standards was kept to a minimum for a few additional reasons. First, we concur with others that the protection and restoration of healthy ecosystem components and functions is the most effective and robust approach to protecting and improving aquatic habitats. While this is clearly a desirable approach, it is not a simple matter given ecosystem complexity. Adequate protection of habitat through protection of ecosystem functions presumes that ecosystem function and response are thoroughly understood. It can only be effective *where it is possible to fully identify all of the ecosystem functions needed to maintain and restore good habitat conditions, as well as the structural and geographic components that contribute to the ecosystem function in question*. This is not always possible given ecosystem complexity and limited information on ecosystem function. Habitat conditions affecting salmon survival are the bottom line and should be monitored to assure that the complex "functional" aspects of the ecosystems needed to maintain good habitat conditions have been adequately identified and protected. Monitoring of condition and trends is especially important given known linkages between habitat condition and survival and the uncertainty over the effectiveness of protection and restoration measures, including the protection of ecosystem functions over time.

Second, numeric standards for some habitat variables can potentially lead to homogenization of habitat conditions. For example, standards for riparian vegetation composition can lead to the "pigeonholing" of vegetation composition. A more sound management approach is, where possible, to fully protect these components of ecosystems and allow them to recover to potential via passive restoration.

Third, application of quantitative standards sometimes courts interventionist approaches aimed at dealing with specific symptoms rather than causes of habitat degradation, e.g. engineered pools. Such approaches are usually unsuccessful in improving habitat conditions and are certainly not sustainable.

Fourth, some uncertainty remains regarding the complete spatial and temporal habitat requirements of salmon. Not every aspect of the biological habitat requirements of the salmon is known in sufficient detail to develop precise quantifiable targets and associated management guidelines. Nonetheless, some relationships between habitat conditions and salmon survival are sufficiently strong that standards must be developed in order to ensure that improvement occurs, unimpeded by land management activities, in highly degraded systems, based on the needs of salmon.

Fifth, the biologically-based standards were kept to a minimum to reduce monitoring and analysis efforts and expedite the completion of screening efforts at the watershed scale. Again, we stress that exclusion of parameters from the core set of habitat standards does not necessarily reflect on their utility for monitoring purposes. Other parameters can and should be monitored.

The biologically-based habitat standards were not stratified by land or channel types because available information does not indicate that the *response of salmon to specific habitat conditions* significantly varies regionally by land or channel types. While stratification issues may be salient to questions of attainability and channel response to perturbation, they are not salient to identifying the habitat conditions needed by salmon to survive. Meeting the habitat requirements of salmon species must be the biological bottom line of efforts to protect and restore spawning and rearing habitat consistent with efforts to stabilize and restore the listed salmon runs.

The recommended biologically-based habitat standards may not be attainable in all systems, even under natural conditions. In systems where recommended habitat standards are not attainable, it is critical that existing conditions not be exacerbated by land use activities, because it may contribute to the extirpation of salmon species from marginal and sensitive habitats. In some sensitive watersheds, such as those in the highly erosive Idaho batholith, tolerable salmon habitat conditions and significant land disturbance are probably mutually exclusive.

It is expected that natural disturbances and processes, such as fire, floods, and droughts may contribute to departures from the habitat standards that may temporarily reduce salmon survival.

Because such departures can affect salmon survival, it is critical that efforts be taken to reduce the anthropogenic contributions that increase the magnitude or duration of the departures from standards. Watershed conditions have a significant influence on how natural disturbance processes affect habitat conditions. In undegraded watersheds, natural disturbance events, such as floods, can have either negligible or even beneficial effects on salmon habitat conditions. In contrast, even minor floods in damaged watersheds can have significant adverse effects on habitat conditions and contribute to declines in salmon populations. This situation underscores the importance of applying adequate watershed protection standards aimed at reducing anthropogenic contributions to habitat degradation and reducing anthropogenic stresses in response to the combined effects of natural- and management-induced change in habitat conditions and salmon survival. Habitat conditions and resultant salmon survival integrate both natural- and management-induced effects. Thus, both types of effects must be considered if improvement in habitat and survival is to be realized. Application of biologically-based standards addresses combined effects by requiring that land management not exacerbate the duration or intensity of habitat conditions that reduce salmon survival and production.

The recent history of land management on federal lands more than adequately indicates that quantitative habitat standards, alone, do not necessarily prompt the types of land management needed to meet standards. Clear, accountable land use standards (or prescriptions) are also needed to ensure that only actions that are consistent with meeting habitat standards are allowed, and that activities inconsistent with meeting those standards are prohibited. For these reasons, we also developed companion land use standards to complement some of the recommended habitat standards.

Alternatives to a biologically-based approach have been proposed for the development of standards for salmon habitat variables. The utility of developing habitat standards based on ranges of natural variation is discussed and evaluated in section 1.5.

1.1 CHANNEL SUBSTRATE: FINE SEDIMENT AND COBBLE EMBEDDEDNESS

1.1.1 Effects on salmon: Incubating eggs and rearing fry both require channel substrates that are relatively free of fine sediment (Everest et al., 1985; Bjornn and Reiser, 1991). Studies have repeatedly documented that increases in fine sediment in streams reduce salmonid survival, production and/or carrying capacity; salmonid populations are typically negatively correlated with the amount of fine sediment in stream substrate (Iwamota et al., 1978; USFS, 1983; Alexander and Hansen, 1986; Everest et al., 1987; Chapman and McLeod, 1987; Rinne, 1990; Hicks et al., 1991; Bjornn and Reiser, 1991; Scully and Petrosky, 1991; Rich et al., 1992; Rich and Petrosky, 1994). The negative correlation of salmonid survival and production to fine sediment has been mainly attributed to reduced survival-to-emergence (STE) and the loss of interstitial rearing habitat in channel substrate.

Most lab and field studies have repeatedly indicated that the success of salmonid emergence from redds is reduced significantly as the amount of fine sediment in spawning gravel increases (Iwamota et al., 1978; USFS, 1983; Everest et al., 1987; Everest et al., 1987; Chapman and McLeod, 1987; Hicks et al., 1991; Scully and Petrosky, 1991; Rich et al., 1992; Maret et al., 1993). The reduction in survival-to-emergence (STE) with increased fine sediment has been ascribed primarily to reduced flow of dissolved oxygen to the incubating eggs (Chapman and McLeod, 1987; Maret et al., 1993) or entombment of the emerging alevins within channel substrate.

A wide variety of fine sediment metrics have been examined in attempting to elucidate and express the effect of fine sediment on salmon STE. Generally, attempts at substrate description have followed one of two approaches: the quantification of the proportion of sediment by weight or volume less than a given particle size or a statistical description of the particle size distribution (Young et al., 1991). A review of these substrate descriptors is beyond the scope of this report, but can be found in Chapman and McLeod (1987) and Young et al. (1991). While different substrate descriptors appear to have variable success in explaining the variation in STE of various species of salmonids (Chapman and McLeod, 1987; Young et al., 1991), most studies clearly indicate that salmonid STE decreases as fine sediment increases, regardless of the type of substrate descriptor used (Chapman and McLeod, 1987). In this report, "fine sediment" refers to sediment particles <0.25 in., as defined by USFS (1983).

Reduced STE caused by high levels of fine sediment is probably the greatest single symptom of natal habitat degradation contributing to the decline in salmon populations in the Snake River basin, for two reasons. First, it is a source of *density-independent* mortality that consistently reduces salmon survival and production, even at the low seeding levels existing in many streams. Reduced STE is a mortality source that is in addition to high levels of downstream mortality at hydroelectric facilities. Density-independent mortality in natal habitat combined with the density-independent impacts to populations at the dams progressively reduces seeding of the natal habitat. Second, as will be discussed, high levels of fine sediment are a pervasive problem throughout natal habitat for spring and summer chinook in the Snake River basin, outside of relatively unperturbed areas.

The consistent effect of reduced STE in streams with high levels of fine sediment is obvious in Figures 1 and 2; salmon densities are consistently several times higher in streams with lower levels of fine sediment than in adjacent streams with high levels of fine sediment, even at extremely low seedings above the mainstem hydroelectric dams (Rich et al., 1992; Rich and Petrosky, 1994). The consistently large differences in juvenile salmon density among streams with different levels of fine sediment as found by Rich et al. (1992) may be due to the rapid emigration of juvenile salmon post-emergence from habitats degraded by fine sediment to habitats with less fine sediment (C. Petrosky, IDFG Fisheries Staff Bio., pers. comm., 1994), which underscores the effect of fine sediment on salmon production. These data clearly indicate that fine sediment conditions in natal habitat profoundly affect salmon survival and production even at existing low seeding levels. It also indicates that these effects are density-independent, occur over a wide range of escapements, and are additive to downstream mortality at hydroelectric facilities. Data from Idaho consistently indicate that tributary streams with higher levels of fine sediment have lower salmon densities, lower salmon survival, and have undergone steeper population declines than in nearby tributaries within the same watersheds with lower sediment levels (Bjornn, 1971a as cited in Seyedbagheri et al., 1987; Platts et al. 1989; Scully and Petrosky, 1991; Rich et al., 1992; Boise National Forest, 1993; Rich and Petrosky, 1994; Petrosky and Schaller, *in press*).

Populations of resident salmonids have also declined significantly in highly sedimented tributary streams of the Salmon River (Platts, 1974a as cited in Seyedbagheri et al., 1987; Boise National Forest, 1993). This decline in resident salmonids strongly corroborates the adverse effects of high fine sediment levels on anadromous salmonids, because resident salmonids are strongly affected by fine sediment, but are unaffected by downstream dams.

The reduction in STE with increased fine sediment appears to be an extremely significant source of increased salmon mortality in many Snake River Basin salmon habitats. Data from the Middle Fork of the Salmon River indicate that the egg-to-parr survival of salmon in streams with high levels of fine sediment is only about 3% while it averages about 29% in nearby streams with lower levels of fine sediment (Scully and Petrosky, 1991 (See Figure 1). Thus, high levels of fine sediment in some Middle Fork streams may have reduced salmon STE by about 90% relative to natural conditions, although these estimates of egg-to-parr survival may not be due solely to reduced STE; they may also reflect the rapid emigration of parr from streams with high levels of fine sediment to streams with low levels of fine sediment (C. Petrosky, IDFG Fish. Staff Bio., pers. comm., 1994). It is estimated that salmon STE remains low in the South Fork of the Salmon River, (R. Thurow, USFS Intermountain Research Station Fish. Bio., pers. comm. 1994), a stream still degraded by fine sediment more than 25 years after catastrophic degradation from landslides from logging roads (Platts et al., 1989). Analysis of spawner-recruitment data from the South Fork of the Salmon River also indicates that salmon survival in natal habitat declined significantly after the landsliding and contributed to declines in the summer chinook salmon populations (Petrosky and Schaller, *in press*).

It also appears that salmon STE is significantly reduced in the Grande Ronde River due to the deposition fine sediment into redds. Purser and Rhodes (*in process*) documented that chronic sedimentation of redds consistently occurs in the Grande Ronde River during the incubation period in streams with >20-30% surface sands. Cleaned gravels, devoid of fines, placed in artificial redds in autumn had >60% fine sediment by depth at about the expected time of salmon emergence after overwintering (Purser and Rhodes, *in process*). At the levels of fine sediment found by Purser and Rhodes (*in process*), existing models and studies (USFS, 1983; Scully and Petrosky, 1991) indicate spring chinook STE is extremely low. While fine sediment clearly reduces salmon STE, no level of fine sediment has been identified below which there is not a negative effect on salmon STE.

It is extremely likely that smaller, geographically-isolated spawning populations of spring and summer chinook will be extirpated within the next several years from streams with high levels of fine sediment. For instance, salmon populations of about 50 females can be expected to be extirpated within about 3 salmon "generations" (about 15 years) when egg-to-fry survival is at less than 20% with existing levels of smolt to adult survival (about 0.4%) according to some simple modeling done by the USFS (1993a (See Figure 3)). Even when smolt-to-adult survival is increased, low survival in natal habitat due to density-independent sedimentation effects is likely to maintain a declining trend in depressed salmon populations. Based on available data, many managed streams have elevated fine sediment levels that have reduced salmon STE. Therefore, it would appear that fine sediment levels in most drainages are accelerating the rate of salmon extirpation. Salmon STE must be increased in degraded drainages in concert with increases in smolt-to-adult survival, if the salmon are to continue to exist.

While salmon actively clean redds of fine sediment during spawning (Everest et al., 1987; Chapman and McLeod, 1987), subsequent sedimentation in the redds by fine sediments is highly likely during the incubation period, especially when ambient surface fine sediment levels are high. This occurs for several reasons. First, sediment transport of fine sediments over redds is likely during the winter period when fine sediment occurs at the surface of the streambed. At a given stream cross section, the discharge required to entrain and transport fine sediment decreases with decreasing sediment size, (Dunne and Leopold, 1978; Richards, 1982; Lisle, 1982; Carson and Griffiths, 1987). On average, the lower the flow magnitude, the greater the duration and frequency that it is exceeded during a given period (Dunne and Leopold, 1978). Therefore, as surface sediment sizes decrease, the flow magnitude needed to transport these particles decreases, which, in turn, increases the probable duration and frequency of the exceedance of the flow magnitude needed to initiate sediment transport, and, hence, duration and frequency of fine sediment transport during incubation periods. Further, shifts in channel morphology caused by high levels of sediment delivery or riparian perturbations tend to increase the ability of a stream to transport sediment at a given flow (Howard, 1987; Carson and Griffiths, 1987; Lisle and Hilton, 1992); this effect also tends to increase the duration and frequency of sediment transport (Lisle and Hilton, 1992). It appears winter flows in salmon habitats in the Snake River Basin are capable of transporting fine sediment during the winter incubation period because increases in fine sediment in cleaned gravels during the spring chinook incubation period have been documented in the South Fork of the Salmon River (King and Thurow, 1991) and the tributaries of the Grande Ronde River (Purser and Rhodes, *in process*).

Second, fine sediment rapidly infiltrates into clean gravels once the transport of fine sediment occurs (Carling, 1984; Diplas, 1991). It appears that the intrusion of fine sediment into coarser bed substrate occurs even at very low sediment concentrations in flow (Diplas, 1991). Intrusion of fine sediment into clean gravels ceases only if sediment transport ceases (Diplas, 1991), a surface seal of fine sediment forms (Lisle, 1989), or all interstitial spaces are filled (Diplas, 1991). Lisle (1989) noted that sedimentation in cleaned gravels (such as redds) appears to be inevitable once bedload movement of fine sediment occurs. Deposition of fine sediment as surface seals has been noted in several studies of sediment conditions in artificial redds (Lisle, 1989; Maret et al., 1993) and/or cleaned gravel substrates (King and Thurow, 1991; King et al., 1992). Notably, surface seals are expected to be as effective in reducing dissolved oxygen supply to incubating eggs as intrusion of sediment into interstitial spaces at depth because flow rates through layered porous media are limited by the least permeable layer (Freeze and Cherry, 1979). It appears that the dissolved oxygen supply to incubating eggs decreases as flow rates through redds decrease (Chapman and McLeod, 1987). Higher flows subsequent to sedimentation of redds may scour away fine sediment at the surface layer of the redd (Lisle, 1989), but removal of intruded fine sediment at depth appears to require flows that would entrain all the sediment particles in the channel substrate at depth in the bed (Diplas, 1991); this would probably scour redds.

Field investigations in Snake River Basin salmon habitats indicate that the sedimentation of redds by fine sediment is common. Available data from the Snake River Basin consistently indicate that cleaned gravels or gravels in artificial redds show an increase in fine sediment levels during the incubation period (King and Thurow, 1991; King et al., 1992; Maret et al., 1993; Purser and Rhodes, in process; (See also: Figure 4)), except where it appears interstitial spaces were already completely filled with fine sediment (King and Thurow, 1991). Monitoring in the Upper Grande Ronde River indicates significant intrusion of fine sediment into cleaned gravels in artificial redds without the development of a surface seal; visual observations indicate that sedimentation ceased only when the interstitial spaces between gravels had been completely filled by fine sediment (Purser and Rhodes, *in process*). Chronic sedimentation of artificial redds in the Grande Ronde River tributaries occurred in streams where surface fine sediment was greater than 20% (Purser and Rhodes, *in process*). Given both available field evidence and sediment transport mechanics, it is expected that the cleaned gravels in redds persist only for a short time in streams with high sediment loads or high levels of fine sediment at the substrate surface because the fine sediment is easily transported and sediment conditions in the cleaned gravels rapidly respond to bedload sediment transport (Lisle, 1989; Diplas, 1991). Thus, the active cleaning of gravels within redds by spawning salmon probably has a limited effect on the ultimate survival-to-emergence of salmon because it probably cannot offset rapid subsequent sedimentation by fine sediment, especially in streams with fine sediment at the bed surface.

Shifts towards reduced particle sizes in spawning substrate renders salmon redds more susceptible to scour, because smaller particles can be entrained by low flows that are frequently exceeded (Howard, 1987). Scouring of spawning reaches has been documented in rivers in coastal streams with rain-dominated hydrology. Nawa and Frissell (1993) found that channel scour in spawning reaches was greater and more frequent in basins with generally higher sediment loads in coastal Oregon. Lisle (1989) documented that scouring of artificial redds also occurred during the incubation period in northern California. However, the salmon habitats of the Snake River Basin have hydrology dominated by snowmelt and generally experience much lower and less flashy winter streamflow than coastal streams during the incubation period.

There has been limited documentation of the scouring of redds within the Snake River Basin. Monitoring has indicated some occurrences of scouring in the Blue Mountains in areas that may be representative of spawning reaches (M. Purser, Conf. Tribes of the Umatilla Indian Reservation Hydrologist, pers. comm., 1993). Maret et al. (1993) found that about 2 out of 9 artificial redds in stream reaches affected by disturbance were partially scoured out during the incubation period in an investigation of redd dynamics of brown trout in Idaho. None of the artificial redds were scoured in the less impacted reaches (Maret et al., 1993). However, available data appear to indicate that sedimentation of redds, rather than scouring, is the more common consequence of increased fine sediment in salmon habitats in the Snake River Basin (King and Thurow, 1991; King et al., 1992; Purser and Rhodes, *in process*). The effect of redd scouring on salmon survival is probably density-independent.

Increased levels of fine sediment in channel substrate reduce the interstitial space in stream substrate required by juvenile salmon during the rearing lifestage (Bjornn, 1971b; Chapman and Bjornn, 1969; Bustard and Narver, 1975a). Cobble embeddedness is a measure of the amount of interstitial space for rearing fry (Harvey, 1989). The density of rearing anadromous salmonids in habitat is typically reduced as cobble embeddedness is increased (USFS, 1983; Chapman and McLeod, 1987; Bjornn and Reiser, 1991). Cobble embeddedness levels of 50 to 60% result in the elimination of fry from the habitat (Chapman and McLeod, 1987) and the food base is altered when cobble embeddedness is in the range of 70% (Bjornn et al., 1977, as cited by Chapman and McLeod, 1987). Some researchers have opined that winter interstitial space may be likely factors limiting carrying capacity of juvenile salmon in natal habitat throughout much of the Idaho batholith (Chapman and McLeod, 1987). Increased cobble embeddedness reduces available winter interstitial habitat and may, consequently, increase pre-smolt mortality during the winter. No level of cobble embeddedness has been established beneath which the capacity of winter rearing habitat is unaffected (USFS, 1983; Harvey, 1989).

Interstitial space is a critical requirement of rearing salmon during winter as indicated by the best available data (Chapman and McLeod, 1987). Interstitial space is of great concern because winter habitat requirements are generally more narrow and restrictive for salmonids in stream environments than those in other seasons (McMahon and Hartman, 1989). Winter salmonid positions are typically in areas with slower water velocity, deeper water, and/or greater cover than in summer (Heifetz et al. 1986; Cunjak and Power, 1986; Riehle and Griffith, 1993). Interstitial habitat in the substrate meets these requirements. The harsh conditions of winter make salmonid survival generally low and highly variable (Kiefer and Forster, 1991). Chapman and Witty (1993) have postulated that this low survival could be attributed to the degradation of winter habitat by accelerated sedimentation. Winter survival has frequently been considered to be a critical bottleneck in production of salmonids in many regions (Nickelson et al., 1992a and b; Heifetz et al. 1986; Mason 1976; Dolloff, 1987).

Monitoring data from the Snake River Basin indicate that salmon density decreases as cobble embeddedness increases and interstitial space is reduced. Espinosa (unpublished data) monitored the amount of "free winter particle" in clean rubble emplaced in streams as a measure of available interstitial space in the rubble over time. Free winter particle is an indicator of changes in embeddedness in that a decrease in the amount of free winter particle equates to an increase in cobble embeddedness. He found that in a heavily sedimented stream (Lolo Creek; cobble embeddedness=50%) that a 50% decrease in free winter particle over time (See Figure 4) was accompanied by a 71% decrease in winter density of juvenile salmon.

Other data appear to indicate that reductions in cobble embeddedness and attendant increases in interstitial space are correlated by increases in salmon density. Hillman et al. (1987) found that 80% of rearing chinook in Red River, Idaho, a highly sedimented tributary of the Clearwater River, emigrated in October when stream temperatures ranged from about 39-46°F, apparently because winter habitat conditions were inhospitable due to low availability of interstitial space (high cobble embeddedness). Fish that remained to overwinter selected undercut banks with overhanging sedges and grasses for cover and with water velocity less than 0.4 ft/s. After cobbles were placed under banks and in mid-channel, chinook used these microhabitats extensively (up to 19 times higher densities than sites without interstitial space among cobbles) until they became too embedded with fine sediment over time.

Bjornn (1971b) noted that the tendency to leave two Idaho stream sites before onset of winter was largely determined by the presence of suitable, unembedded cobble substrate. He found that trout and salmon left reaches with only fine substrates and took up residence in reaches with unembedded rubble. Rubble habitats provide a large measure of physical protection to fish hiding in the substrate from scouring by ice and dislodgement of relatively inactive fish in cold water by higher flows. Additions of rubble to sections of a stream with fine substrate induced 1+ age steelhead to overwinter (Bjornn, 1971b). The same behavior was observed in laboratory streams: approximately twice as many steelhead and chinook emigrated from laboratory streams with gravel substrate as emigrated from laboratory streams with unembedded rubble during simulated winter conditions.

These studies and others indicate that rearing salmon readily use interstitial spaces in substrate when they are available and emigrate from highly embedded areas, especially during harsh winter conditions (Chapman and McLeod, 1987). While these studies indicate that additions of rubble to highly embedded streams can increase salmon density and provide important interstitial habitat during winter, the benefits to salmon are extremely short-lived because the rubble and interstitial space is rapidly inundated by sediment in areas where fine sediment levels are high.

Increased transport of fine sediment also leads to deleterious channel change, such as pool in-filling (Jackson and Beschta, 1984; Alexander and Hansen, 1986; Lisle and Hilton, 1992; MacDonald et al., 1991; McIntosh, 1992). Pool loss reduces the carrying capacity of fish habitat and often reduces salmonid production (Alexander and Hansen, 1986; Fausch and Northcote, 1992; Nickelson et al., 1992a). Streams with abundant fine sediment also typically widen over time which can exacerbate seasonal temperature extremes by increasing the stream surface area at all discharge levels. For instance, Alexander and Hansen (1986) added fine sediment to a stream in Michigan and found that the channel became wider and shallower; summer water temperatures exhibited a statistically significant ($p<0.05$) increase of about 2.7°F relative to the pre-treatment period.

Increases in fine sediment in stream systems have multiple effects on salmon habitat that can synergistically reduce salmon survival and production. High levels of fine sediment not only reduce STE, they also reduce summer and winter rearing habitat by reducing pool volumes and interstitial rearing space. The effects of loss of rearing habitat on salmon survival and production can be exacerbated by the effect of channel change on water temperature. High levels of fine sediment also alter the food web. The work of Alexander and Hansen (1986) provides a picture of the multiplicity of effects of increased levels of fine sediment on salmonid production that may be generally applicable to Snake River Basin salmon habitats even though it focused on the response of brook trout in Michigan. After adding fine sediment to a stream, the channel widened and became shallower, most pools were eliminated (which reduced effective cover), summer water temperature increased, and the abundance of food was reduced drastically. In response to these changes, brook trout populations dropped to half of their normal abundance during the pre-treatment period in the reaches with added sediment, apparently due to greatly decreased survival at the egg-to-fry and fry-to-fingerling lifestages (Alexander and Hansen, 1986). This work underscores that high levels of fine sediment in salmonid habitats do not have single effects at a single lifestage, but have multiple negative effects that probably interact synergistically to reduce salmon survival.

1.1.2 Activities affecting stream substrate/fine sediment: Fine sediment and cobble embeddedness levels are strongly controlled by sediment delivery. Increases in fine sediment and cobble embeddedness are generally caused by increases in sediment loading to stream channels in excess of the streams' ability to transport sediment (USFS, 1983; Howard, 1987; Platts et al, 1989; Diplas, 1991; Chamberlin et al., 1991). It is well-documented that increases in sediment delivery can increase the amount of fine sediment present in stream substrate (Dunne and Leopold, 1978; Richards, 1982; Megahan, 1984a; Howard, 1987; Swanson et al., 1987; Platts et al., 1989; Dietrich et al., 1989; MacDonald and Ritland, 1989; Rinne, 1990; Diplas, 1991; MacDonald et al., 1991; Hicks et al., 1991, Chamberlin et al., 1991; Swanston, 1991; Lisle and Hilton, 1992). It is also known that activities that remove vegetation, compact and disrupt soils, and/or increase overland flow within watersheds increase erosion and, hence, are likely to cause increased sediment loading (Dunne and Leopold, 1978; USFS, 1980; Swanson et al., 1987; Everest et al., 1987; MacDonald et al., 1991). Studies in the Idaho batholith indicate that streams in pristine basins with natural sediment loads have significantly lower fine sediment and cobble embeddedness than in managed basins (Burns, 1984; Boise National Forest, 1993; Overton et al., 1993). The same pattern has also been observed in streams in the Blue Mountains of Oregon (J. Rhodes, unpublished field notes, 1993).

In the Snake River Basin, agriculture is probably the single largest sediment source (Idaho Dept. of Health and Welfare, 1989). Sediment delivery from agriculture is the largest cause of beneficial use impairment in Idaho (Idaho Dept. of Health and Welfare, 1989). Sediment delivery from agriculture is one of the most significant impacts affecting fish habitat in the Tucannon (Theurer et al., 1985) and the Grande Ronde (ODEQ, 1989, ODFW et al., 1990). Tilled agriculture can increase erosion by more than 100 times natural rates on a per unit basis, although the rate of increase varies greatly with type of treatment, crop type, climate, soil type, and topography (Dunne and Leopold, 1978; USFS, 1980).

Grazing significantly increases sediment delivery (Lusby, 1970; Dunne and Leopold, 1978; MacDonald et al., 1991; Platts, 1991; Boise National Forest, 1993) and is a major nonpoint source of sediment in the Snake River Basin (Idaho Dept. of Health and Welfare, 1989). Lusby (1970) found that a grazed watershed in Colorado yielded about 1.8 times the amount of sediment as an ungrazed watershed. In Bear Valley Creek in Idaho, it is estimated that grazing has increased sediment delivery to the stream to about 60% over natural (Boise National Forest, 1993). Sediment delivery from grazing is extremely high because the impacts are concentrated in riparian areas (Platts and Nelson, 1985; Kauffman et al., 1983; Ohmart and Andersen, 1986, MacDonald et al., 1991; Platts, 1991). Virtually every effect of grazing on soils, vegetation, hydrology, and streambanks greatly increases channel erosion and subsequent sedimentation (Parsons, 1965; Graf, 1979; Platts, 1981b; Kauffman et al., 1983; Harvey and Watson, 1986; MacDonald et al., 1991; Platts, 1991). In the Middle Fork Salmon River, fine sediment and sediment delivery are much higher in streams draining grazed watersheds than in streams draining ungrazed watersheds, resulting in depressed salmon survival in grazed watersheds (Scully and Petrosky, 1991; Rich et al., 1992; Boise National Forest, 1993 (See Figures 2 and 5)). In these streams, fine sediment levels are inversely correlated with bank stability (Figure 6); the loss of bank stability caused by grazing may be a major cause of elevated fine sediment levels. Grazed areas generally have more fine sediment than similar ungrazed areas (Platts, 1991).

Mining activities can cause significant increases in sediment delivery. While mining may not be as geographically pervasive as other sediment-producing activities, surface mining typically increases sediment delivery much more per unit of disturbed area than other activities (Dunne and Leopold, 1978; USFS, 1980; Richards, 1982; Nelson et al. 1991) due to the level of disruption of soils, topography, and vegetation. Relatively small amounts of mining can increase sediment delivery significantly. For instance, in Bear Valley Creek in Idaho, it is estimated that mining contributed more than 17 million ft^3 of sediment to the stream system after the stream cut through tailings (Nelson et al., 1991). It is estimated that sediment contributions from historic mining in the watershed of Bear Valley Creek continue to elevate sediment delivery to the stream at about 1.5 times natural levels (Boise National Forest, 1993) more than 20 years after the cessation of mining (Nelson et al., 1991). Surface fine sediment and cobble embeddedness in Bear Valley Creek are extremely high, averaging about 56% and 70%, respectively (Boise National Forest, 1993); grazing has also contributed to these high levels of fine sediment. Fine sediment or cobble embeddedness levels have been documented to be higher downstream of mines than upstream (Chapman and McLeod, 1987; Nelson et al., 1991).

Logging and logging roads typically accelerate sediment delivery to streams on the order of two to ten times natural rates (Geppert et al., 1984; MacDonald and Ritland, 1989; Chamberlin et al., 1991). Logged areas contribute significant quantities of sediment to streams, especially in steep and/or erosive terrain or where proximate to streams (Everest et al., 1987; Swanson et al., 1987; Chamberlin et al., 1991; Hicks et al., 1991). In Idaho, ground-cable logged areas erode at about 1.6 times natural rates per unit area, on average, over a six year period following logging (King, 1993). Logging methods creating less soil and vegetation disturbance cause lower increases in erosion and resultant sediment delivery (USFS, 1981). Logging can also increases the rate of mass erosion (Swanston, 1991; Chamberlin et al., 1991). Riparian zone logging can increase erosion and sediment delivery in a number of ways: 1) increased fluvial channel erosion due to reductions in bank stability from vegetation (Parsons, 1965; Graf, 1979; Harvey and Watson, 1986; Chamberlin et al., 1991; Hicks et al., 1991); 2) increased frequency of mass failures (Megahan et al., 1978; Gray and Megahan, 1981; Iverson and Major, 1986; Megahan and Bohn, 1989); and, 3) increased sediment transport due to the loss of sediment storage behind downed wood (Beschta, 1979; Megahan, 1982, Heede, 1985; MacDonald and Ritland, 1989; Swanston, 1991). However, the majority of sediment delivered from logging activities is from roads and road construction (Megahan et al., 1978; Dunne and Leopold, 1978; Geppert et al., 1984; MacDonald and Ritland, 1989; Chamberlin et al., 1991; Furniss et al., 1991). Roads in the Idaho batholith have been shown to increase surface erosion by 220 times natural rates on a per unit area basis (King, 1993). However, roads also greatly increase mass erosion rates. Megahan et al. (1978) found that 88% of landslides inventoried in Idaho were associated with roads. Other studies have repeatedly demonstrated that roads greatly increase the frequency of landslides, debris flow, and other types of mass erosion (Megahan et al., 1978; Dunne and Leopold, 1978; Geppert et al., 1984; MacDonald and Ritland, 1989; Chamberlin et al., 1991; Furniss et al., 1991). It appears that logging always causes some increase in sediment delivery to streams even when low impact logging systems are used in conjunction with vegetative buffers (Megahan, 1987; Heede, 1991) or existing BMPs are stringently implemented (Lynch and Corbett, 1990). Studies have repeatedly shown that logging and related activities significantly increase fine sediment in downstream fish habitat (Geppert et al., 1984; Everest et al., 1985; Everest et al., 1987; MacDonald and Ritland, 1989; Platts et al., 1989; MacDonald et al., 1991; Chamberlin et al., 1991; Hicks et al., 1991).

Available data indicate that fine sediment levels in many managed drainages have been elevated in both the short term (Clearwater National Forest, 1993) and long term (Platts et al., 1989; McIntosh, 1992; Boise National Forest, 1993)(See Figures 7, 8, and 9). Cobble embeddedness in many managed watersheds is extremely elevated and appears to range from about 30-90% (Idaho Dept. of Health and Welfare, 1991; Clearwater National Forest, 1991a; Wallowa-Whitman National Forest, 1992; Boise National Forest, 1993). Many streams with elevated levels of fine sediment have not shown significant improvement (Figure 7); some have continued to deteriorate (Figure 8) (Clearwater National Forest, 1993). With existing levels of sediment delivery and watershed conditions (Boise National Forest, 1993; Clearwater National Forest, 1991a) it is likely that levels of fine sediment will either increase or be maintained at adversely high levels unless sediment delivery is reduced to below the transport capacity of streams.

Studies on the South Fork of the Salmon River indicate that the recovery of degraded stream substrate conditions is contingent on reducing sediment loads (Platts et al., 1989; Megahan et al., 1992; Bohn and Megahan, 1991; Idaho Dept. of Health and Welfare, 1991). High levels of sediment delivery maintain adversely high levels of fine sediment and cobble embeddedness (Platts et al., 1989; Megahan et al., 1992; Idaho Dept. of Health and Welfare, 1991; Diplas, 1991; Swanston, 1991). Data from the South Fork of the Salmon River indicate that, after its catastrophic sedimentation, no appreciable reduction in fine sediment occurred until sediment delivery was reduced to less than 25% over natural (Megahan et al., 1992 (See Figure 9)). Recovery of substrate conditions in the South Fork Salmon River has currently ceased (Platts et al., 1989) and it appears that fine sediment levels are now in quasi-equilibrium with current sediment loads from roads and other sources (Platts et al., 1989; Idaho Dept. of Health and Welfare, 1991). Currently, it is estimated that sediment delivery in the South Fork of the Salmon River is at about 18% over natural (Idaho Dept. of Health and Welfare, 1991), although there is considerable potential for error in such estimates (King, 1993). Trend data from streams in the Idaho batholith indicate that high levels of substrate fine sediment are either maintained or increased under sustained sediment loads estimated to be more than 20% over natural ((See Figures 4, 7, and 8)). Although the data are fragmentary and have potential sources of error, it appears that <u>total</u> sediment delivery in most basins will have to be reduced to less than 20% over natural if fine sediment conditions in degraded streams can be expected to improve.

To our knowledge, the South Fork Salmon River is the only managed stream that has been documented to have undergone significant improvement after it was catastrophically degraded by massive sediment deposition (Platts et al., 1989; Megahan et al., 1992). While it improved dramatically from the late 1960s to 1980 in response to reduced sediment delivery caused by a logging moratorium, it ceased improving in 1980 and still has not fully recovered from the catastrophic sedimentation (Platts et al., 1989; Idaho Dept. of Health and Welfare, 1991; Megahan et al., 1992). It is unlikely that fine sediment conditions in the South Fork Salmon River will show future improvement unless sediment delivery from anthropogenic sources is reduced further (Platts et al., 1989; Idaho Dept. of Health and Welfare, 1991; Megahan et al., 1992). The South Fork Salmon River data (Platts et al., 1989; Idaho Dept. of Health and Welfare, 1991; Megahan et al., 1992) and other available models and information (Scully and Petrosky, 1991) indicate that the reductions in fine sediment in the South Fork Salmon River may have increased STE by **an order of magnitude** over 15 years, although salmon STE in the South Fork Salmon River appears to continue to be depressed at low levels (R. Thurow, USFS Intermountain Research Station Fish. Bio., pers. comm., 1994). The case history of South Fork Salmon River clearly indicates that salmon survival can be significantly increased in a fairly short time period, but only if the causes of degradation are arrested and reversed.

While limited data indicate that there are some limited prospects for improvement in substrate conditions and salmon survival at sediment delivery levels less than 20% over natural, the actual level of sediment delivery needed for recovery will vary among streams and is dependent on local hydrology, climate, vegetation, and channel morphology. However, it is unlikely that any degraded streams can fully recover from sediment degradation as long as sediment delivery remains elevated. Historically, these stream segments were probably in some form of quasi-equilibrium with natural cycles of sediment loading and discharge. The listed salmon spawn and rear in stream and valley environments that are strongly depositional in nature. Elevation of sediment delivery under such conditions inexorably leads to increased fine sediments (Richards, 1982; Howard, 1987; Carson and Griffiths, 1987; Dietrich et al., 1989; Diplas, 1991; Lisle and Hilton, 1992). The notion that there is "excess sediment transport capacity" in stream reaches that provide spawning and rearing habitat in the Snake River Basin is probably spurious. In some streams, relatively small increases in anthropogenic sediment delivery (<10% over natural) can shift substrate conditions towards smaller particle sizes (D. Burns, Payette National Forest Fish. Bio., pers. comm., 1993).

Shifts toward smaller sediment sizes in bedload transport also decrease the sediment size of substrate (Dietrich et al., 1989). Increased erosion in small perennial channels caused by the loss of vegetation and/or increased flows can decrease the particle size of sediment transport downstream because flows in these small streams primarily transports fine sediment due to limited stream competence. For instance, the bulk of sediment transported in some small ephemeral streams in the Grande Ronde River watershed is less than 0.08 in. in diameter (R. Gill, Wallowa-Whitman National Forest District Hydrologist, pers. comm., 1993).

Spawning and rearing habitat in the Snake River Basin is extremely susceptible to increases in fine sediment in response to elevated sediment loads due to the basin's environmental characteristics. Streams are most sensitive to increases in fine sediment in areas that have snowmelt-dominated hydrology, more arid climates, significant mass erosion, granitic geology, low gradient streams, steep terrain, and low frequency of large woody debris (Everest et al., 1987). Notably, many of the Snake River Basin watersheds have just such characteristics.

1.1.3 Evaluation: Studies consistently indicate that salmonid survival decreases as fine sediment in substrate increases. High levels of surface fines can lead to sedimentation of redds, pool loss, and deleterious channel change. A wide variety of metrics have been used to predict salmon survival and have yielded mixed results under varying environmental conditions (Chapman and McLeod, 1987; Young et al., 1991). However, given the goal of improving and protecting salmon survival, it is not necessary to be able to accurately predict salmon survival as a function of fine sediment levels; it is enough to know that salmon survival decreases with increasing fine sediment. If salmon habitat and survival are to be protected and improved, it is apparent that fine sediment levels in degraded streams must be reduced and that increases in fine sediment levels must be avoided in all streams.

Data also indicate that rearing salmon require interstitial space in cobble and rubble substrates especially during the winter (Chapman and McLeod, 1987). As cobble embeddedness increases, interstitial habitat is lost, and the production capability of salmon habitat is probably diminished. Some researchers have opined that cobble embeddedness probably cannot be used to quantitatively predict salmon response, but a high degree of accuracy in predicting of salmon response is not necessary to requiring that cobble embeddedness levels be protected or improved in salmon habitat as part of efforts to increase salmon survival, given available information.

Available information also amply indicates that land disturbing activities cumulatively increase fine sediment levels by increasing on-site erosion and sediment delivery to streams.

Much of the natal habitat for spring and summer chinook salmon streams in the Snake River Basin has been so degraded by sediment contributed by the separate and combined effects of grazing, agriculture, logging, road construction, and mining that they no longer fully support the beneficial uses of salmon spawning and rearing (Theurer et al., 1985; ODEQ, 1989; Idaho Dept. of Health and Welfare, 1989; Scully and Petrosky, 1991; Rich et al., 1992; Wallowa-Whitman National Forest, 1992; Boise National Forest, 1993; Henjum et al., 1994). Elevated sediment delivery caused by logging, logging roads, grazing and mining has elevated fine sediment levels throughout many of the watersheds managed for "multiple uses" in the Snake River Basin, especially within the erosive Idaho batholith (Megahan et al., 1978; Idaho Dept. of Health and Welfare, 1989; Platts et al., 1989; Nez Perce Tribe and IDFG, 1990; Clearwater National Forest, 1991a; Espinosa and Lee, 1991; Idaho Dept. of Health and Welfare, 1991; Scully and Petrosky, 1991; Rich et al., 1992; Boise National Forest, 1993; NMFS, 1993). Studies have consistently indicated that sedimentation of fish habitat in the South Fork of the Salmon River played a role in reducing the populations of anadromous fish (Bjornn, 1971a as cited in Seyedbagheri et al., 1987; Platts et al., 1989; Petrosky and Schaller, *in press*). Similar sediment problems also exist in the Grande Ronde and Tucannon (Theurer et al., 1985; ODEQ, 1989; ODFW et al., 1990; Anderson et al., 1992; Wallowa-Whitman National Forest, 1992).

Based on available data, fine sediment levels range from 20 to 60% in many streams outside of wilderness and roadless areas(Platts et al., 1989; Scully and Petrosky, 1991; Rich et al., 1992; Clearwater National Forest, 1991a and b; Wallowa-Whitman National Forest, 1992; Purser and Rhodes, in process; Boise National Forest, 1993; Clearwater National Forest, 1993). Therefore, it is likely that salmon STE in many watersheds outside of wilderness/roadless areas has been significantly reduced and may average only 10-15% (USFS, 1983; Scully and Petrosky, 1991; Boise National Forest, 1993). Given this estimated survival rate, it is probable that salmon populations will continue to decline in most drainages and that widespread extirpations can be expected over the next two decades unless fine sediment levels are reduced (See Figure 3). Interstitial rearing habitat has also been reduced in many managed drainages due to high levels of cobble embeddedness. It will be necessary to abate sediment delivery from nonpoint sources within watersheds to restore substrate conditions that will allow salmon to continue to persist.

Substrate conditions are strongly influenced by sediment delivery and transport processes that operate at the watershed scale (Dietrich et al., 1989; Chamberlin et al., 1991; Swanston, 1991; Hicks et al., 1991). Because land disturbing activities increase sediment delivery and lead to increased fine sediment and cobble embeddedness, it will be necessary to limit the types, magnitude and locations of human disturbance within watersheds, if fine sediment and cobble embeddedness are to be at levels that allow spawning salmon populations to persist, much less increase. However, interactions among natural watershed processes, climatic events, land use, sediment delivery, sediment routing, and resultant substrate conditions are complex and poorly predictable. Given current uncertainties together with what is known about interactions among land uses, sediment delivery, and substrate at the watershed scale, it does not appear that land use standards, alone, can be used as a surrogate to substrate standards to assure that substrate conditions are improved or protected. Although we recommend constraining sediment producing activities and total sediment delivery based on best available information, the recommendation is based on data that are fragmentary and have considerable potential for error. While available information clearly indicates that sediment delivery must be reduced to improve substrate conditions in degraded streams, it is not certain that meeting our recommendations regarding sediment delivery and other land use standards will protect and improve substrate conditions in all cases. Therefore, biologically-based habitat standards for substrate are needed as part of screening for the need to improve existing habitat conditions. Appendix B provides details on blindly relying on assumed sediment delivery thresholds. Given the complexity of watershed-scale sediment dynamics, substrate monitoring is needed to track the effectiveness of combined restoration and protection measures as part of an adaptive management approach.

Although a wide variety of metrics have been used to measure and express fine sediment levels and elucidate the effects on salmon survival with varying levels of success (Everest et al., 1987; Chapman and McLeod, 1987; Bjornn and Reiser, 1991; Young et al., 1991; Peterson et al., 1992), we recommend setting a fine sediment standard in spawning habitat for surface fines as part of the screening process for several reasons. First, high levels of surface fine sediment increase the propensity for the sedimentation of salmon redds leading to reduced STE. Second, available data indicate that surface fine sediment levels are linked to salmon survival and the level of use of habitat (Rich et al., 1992; Rich and Petrosky, 1994). It is, therefore, prudent to require that surface fine sediments be reduced in degraded habitats and that increases be avoided in all habitats. Third, surface sediment conditions ultimately influence the substrate conditions at depth that can also affect salmon STE. Platts et al. (1989) found that trends in surface and subsurface fines were correlated in the South Fork of the Salmon River; subsurface fines at sites were generally higher than subsurface fines. Although surface fine sediment levels may not always accurately indicate trends in fine sediment at depth, it is likely that increases in surface fines lead to increased levels of fine sediment at depth. It is unlikely that degraded substrate conditions can improve unless surface fine sediment conditions first show improvement. Fourth, levels of fine sediment at the substrate surface should exhibit fairly rapid response over time to changes in sediment delivery at the watershed scale. Once combined measures are effective in adequately reducing sediment delivery, surface fine sediment conditions should show more rapid improvement than conditions at depth. Fifth, monitoring of surface fine sediment should be more tractable and require less effort than other measures. This should expedite monitoring efforts and screening at the watershed scale.

However, it should be noted that using surface fine sediment levels as the only measure of substrate condition in habitat has some disadvantages. Fines at depth clearly affect salmon survival. Although surface fine sediment and fine sediment at depth can be significantly correlated (Platts et al., 1989), they may not always be well correlated. Fine sediment levels can increase at depth by intrusion into fairly clean substrate without measurably increasing at the surface. Fine sediment levels can also remain deleteriously high at depth while surface conditions indicate improvement because the removal of intruded fine sediment at depth appears to require flows that are not only low in fine sediment concentration but also large enough to entrain all the sediment particles in the channel substrate at a depth (Diplas, 1991). Thus, improvements in surface fine sediment may not necessarily equate to improvement in substrate conditions at depth. Therefore, although surface fine sediment levels should be adequate for screening for the need to improve habitat conditions, other substrate parameters such as fines by depth should be monitored to assure that habitat conditions improve. Nonetheless, fine sediment by depth is not set as a mandatory element in the screening process.

Interstitial habitats appear to be critically important to salmon in the Snake River Basin, based on best available information. Although cobble embeddedness is a time-consuming measure of the availability of interstitial space in rearing habitat, it remains the most widely used measure (MacDonald et al., 1991). Chapman and McLeod (1987) noted that a conservative approach to the protection of winter habitat, given best available information would be to avoid cobble embeddedness greater than 25% until better information becomes available.

We recommend that watersheds should be managed so that surface fine sediment averages less than 20% in spawning habitat. However, it cannot be assumed that habitats with surface fine sediment averaging less than 20% have not been affected by land use or are not negatively affecting salmon survival and production. Given existing information, any increase in surface fine sediment levels in spawning habitat probably represents a deleterious modification of salmon habitat. Therefore, we recommend that watersheds should be managed so that there is no increase in existing surface fine sediment even when it averages less than 20% in spawning habitat.

We also recommend that watersheds should be managed so that cobble embeddedness averages less than 30% within rearing habitat. We concur with Chapman and McLeod (1987) that incremental increases in fine sediment above current conditions are likely to reduce the overwinter survival of salmonids. Given the importance of interstitial rearing space as habitat requirement of salmon, it is prudent to protect all systems from increases in cobble embeddedness. Therefore, we recommend that where cobble embeddedness is less than 30%, no increase in cobble embeddedness should occur.

A low risk approach to land management should be taken in watersheds where substrate standards are not met because of the pronounced effect of fine sediment conditions on salmon survival and combined uncertainties regarding the effectiveness of efforts to reduce sediment delivery at the watershed scale. Therefore, we recommend that where substrate standards are not met, activities that can potentially forestall improvement in substrate conditions should not be implemented or continued (See Table 1). Activities that increase erosion above natural levels can forestall improvement in substrate conditions. Therefore, the initiation or continuation of activities that increase erosion above natural levels are inconsistent with efforts to improve substrate conditions. We recommend that such activities (See Table 1) should not be initiated nor continued until the substrate standards are met or a statistically significant ($p<0.05$) improving trend is documented through monitoring over at least five years.

It is unlikely that fine sediment and cobble embeddedness conditions can be protected or recovered without adequate control of sediment delivery at the watershed scale. Because sediment delivery at the watershed scale exerts a strong influence on fine sediment and cobble embeddedness conditions, we also recommend that the standards for sediment delivery be included as part of screening activities for their consistency with efforts to improve salmon habitat and survival. (For additional detail see Section 3.2 Sediment Delivery).

We recommend the following standards for watersheds where substrate standards are not met and sediment delivery is estimated to be more than 20% over natural: 1) Reduce sediment delivery through suspension of on-going activities and prohibition of the initiation of activities that can increase erosion over natural levels (See Table 1), and implement active restoration measures (e.g., road obliteration) as needed, until substrate conditions meet standards or a statistically significant ($p<0.05$) improving trend over at least 5 years is documented through monitoring <u>and</u> total sediment delivery from anthropogenic sources is estimated to be less than 20% over natural. 2) If substrate conditions do not meet standards after total sediment delivery is estimated to be less than 20% over natural <u>and</u> substrate conditions have exhibited a statistically significant, improving trend over at least five years, activities that can increase erosion should only be implemented/re-initiated when combined with active and passive restoration measures so that they result in net reductions in sediment delivery until substrate conditions meet standards.

We recommend the following for watersheds where substrate standards are not met but total sediment delivery from all anthropogenic sources is estimated to be less than 20% over natural: 1) Eliminate on-going activities and prohibit activities that can increase erosion over natural levels and implement active restoration measures, such as road obliteration, until substrate conditions meet standards or an statistically significant ($p<0.05$) improving trend over at least 5 years is documented through monitoring. 2) Once substrate conditions have exhibited a statistically significant trend over at least five years, activities that increase erosion should only be implemented/re-initiated when combined with active and passive restoration measures that result in net reductions in sediment delivery, until substrate conditions meet standards.

We recommend the following standards for watersheds where substrate standards are met but sediment delivery is estimated to be more than 20% over natural: Activities that increase erosion over natural levels should only be implemented or continued when combined with active and passive restoration measures that result in net reductions in sediment delivery, until sediment delivery is less than 20% or substrate conditions exhibit a statistically significant ($p<0.05$) improving trend over at least five years as documented by monitoring.

We recommend that "factors of safety" should be incorporated into attempts to reach a net reduction in sediment delivery through active restoration combined with land disturbance. For example, Anderson et al. (1992) recommended that active restoration efforts should offset expected increases in sediment delivery by a factor of three. Due to uncertainties in the effectiveness of active restoration and the estimation of sediment, we believe that the threefold reduction in expected sediment delivery recommended by Anderson et al., (1992) is a prudent, minimum factor of safety regarding efforts to offset sediment delivery of activities by active restoration. We also recommend that active restoration efforts should be completed and shown to be effective prior to initiating or continuing land disturbing activities, when a net reduction in sediment delivery is the goal.

If substrate conditions show statistically significant deterioration ($p<0.40$) over any period longer than one year, activities that increase erosion should be suspended until substrate conditions return to their initial condition prior to deterioration; active restoration measures aimed at reducing sediment loads should also be undertaken in such cases. The level of low level of statistical significance ($p<0.40$) regarding habitat deterioration is recommended as part of a risk adverse approach because of the environmental and economic costs associated with failing to address habitat degradation once it has occurred. (For more detail, see Section 1.6 Notes on Statistical Significance).

We recommend that activities that do not meet these suites of standards should be considered inconsistent with efforts to improve salmon survival in natal habitat.

1.2 CHANNEL MORPHOLOGY

1.2.1 Pools and Large Woody Debris (LWD)

1.2.1.1 Effects on Salmon: Large pools with low water velocity are an essential component of salmon habitat (Everest et al., 1985; Sedell and Everest, 1990; Bjornn and Reiser, 1991; MacDonald et al., 1991). Pools are used for resting by adults prior to spawning and for winter and summer rearing by juvenile salmon (Everest et al., 1985; Sedell and Everest, 1990; Bjornn and Reiser, 1991; Hicks et al., 1991). Pools provide critical flow and thermal refugia during seasonal discharge and temperature extremes (Sedell and Everest, 1990; McIntosh, 1992; Nielsen et al., 1994). Pools are generally the most productive rearing habitat (Everest et al., 1985; Sedell and Everest, 1990; Gregory and Ashkenas, 1990; Bjornn and Reiser, 1991; Nickelson et al., 1992a; Fausch and Northcote, 1992). Salmonid density is positively correlated to pool volume and frequency; pool loss reduces the production capability of salmonid habitat (Everest et al., 1985; Sedell and Everest, 1990; MacDonald et al., 1991; Nickelson et al., 1992a; Fausch and Northcote, 1992). Availability of pools during summer low flow periods can be a limiting factor in survival and production of salmonids (Reeves et al., 1990).

Pools are also important components of winter habitat for overwintering salmonids, together with side channels, sloughs, beaver ponds, coarse rubble and boulders, overhanging banks and associated bank vegetation, upturned roots, and debris accumulations. Important types of pools in winter habitat include alcoves or side pools, as well as deep primary pools. Reductions in the quantity, quality, stability, or frequency of these pool types tend to reduce the overwinter survival of the salmonids that depend upon these habitats. In many forested streams, pool volume, frequency, and quality are associated with the size and frequency of LWD (Everest et al., 1985; Heifetz et al., 1986; Sullivan et al., 1987; Bisson et al., 1987; MacDonald et al., 1991).

Pool loss and channel widening typically render streams more susceptible to heating during the summer and icing during the winter (USFS, 1980; Alexander and Hansen, 1986; Beschta et al., 1987; Everest et al., 1987; MacDonald et al., 1991). Pool loss may also reduce the frequency and volume of thermal refugia. Pool depth and volume may be an important aspect for thermal refugia. Pools may have to be deep enough to intercept colder subsurface flow or large and deep enough to allow thermal stratification of the flowing water (Nielsen et al., 1994). Reductions in pool volume or depth may reduce the effectiveness of pools in providing thermal refugia. Reduced frequency and volumes of pools may subject rearing juveniles to more intense predation in shallow waters devoid of cover and more intense competition for food in limited rearing area.

The spatial organization of macrohabitat types within a stream network has significance in the survival of a chinook population and the age class composition of the population. This includes considerations of the frequency of primary pools, the location of clearcut or altered stream reaches relative to other, more stable reaches, and the spatial location of summer rearing areas relative to overwintering areas. Large numbers of coho have been reported leaving mainstem coastal rivers in autumn to seek shelter in tributary sloughs, beaver ponds, side channels, or "wall base" channels (Bustard and Narver, 1975b; Tschaplinski and Hartman, 1983; Cederholm and Scarlett, 1982) where current velocity and risk of downstream displacement or physical injury from floods is low. In the spring, rearing fish re-enter the mainstem. Survival in off-channel areas can least twice as great as in mainstem habitats during the winter period (Bustard and Narver, 1975b). Although main- and side-channel pools (especially deep, primary pools associated with abundant, stable LWD) can provide the deep, low velocity habitats needed for winter rearing of coho and other salmon species, the loss of these complex primary pools plus and elimination of off-channel rearing areas can create a significant limiting factor in coho production in coastal streams (Nickelson et al., 1992b).

Pool quality is also important to salmon survival. Platts (1974) found a significant positive relationship between pool quality and the standing crop of salmonids. Because the various species and age classes of fish are distributed in streams in relation to macrohabitat type (e.g., riffle, pool), variant of the macrohabitat type (e.g., dammed pool, plunge pool) and also quality (e.g., depth, LWD volume, cobble embeddedness), the productive capability of a stream is dependent on the diversity of habitat qualities by type. Reduction in depth and cover diversity of pools owing to sedimentation and LWD loss limits the capability to support pool dependent species and age classes. LWD serves a key role in increasing the quality of pools by providing hiding cover, slow water refuges, shade, and deep water zones.

LWD is a vital aspect of habitat for spring and summer chinook and other salmonids (Everest et al., 1987; Bisson et al., 1987; Gregory and Ashkenas, 1990; MacDonald et al., 1991; Nickelson et al., 1992a; Fausch and Northcote, 1992). LWD provides a wide variety of functions in salmon habitat including: 1) hiding cover for rearing juveniles and returning adult fish (Everest et al., 1985); 2) structure for the maintenance and development of pools and habitat complexity (Everest et al., 1985; Bisson et al., 1987; Gregory and Ashkenas, 1990); 3) prevention of channel erosion; and 4) sediment storage sites which reduce downstream sediment transport (Beschta, 1979; Megahan, 1982; MacDonald and Ritland, 1989; Gregory and Ashkenas, 1990; Swanston, 1991; MacDonald et al., 1991).

Many studies indicate a strong positive relationship between salmonid production and LWD (Everest et al., 1985; Bisson et al., 1987; MacDonald et al., 1991; Nickelson et al., 1992a; Fausch and Northcote, 1992). Thus far, under natural LWD loadings, an upper end to the relationship between salmon production and LWD has not been found (Bisson et al., 1987). It appears that any reduction in natural LWD recruitment to salmon habitat reduces habitat quality and salmon production (Everest et al., 1985; Alexander and Hansen, 1986; Bisson et al., 1987; MacDonald et al., 1991; Nickelson et al., 1992a; Fausch and Northcote, 1992).

The role of LWD in creating large, deep pool habitats, is a key factor in providing high quality winter habitat. In southwestern British Columbia, Tschaplinski and Hartman (1983) reported that the midwinter numbers of juvenile coho salmon in each study reach were linearly correlated with LWD volume (R^2=0.89); coho juveniles were eliminated from clearcut stream sections without stable LWD when late fall/early winter freshets occurred. Murphy et al., (1984) found a similar relationship in Alaskan streams. Dolloff (1987) found that winter production of coho was lowest in areas with low pool volumes and where LWD was ineffective in creating habitat structure. Heifetz et al. (1986) cite several Alaskan studies emphasizing the importance of maintaining high quality overwintering habitat created by LWD for salmonids that spend one or more years in freshwater. Multiple year freshwater rearing periods are also common for salmon in the Snake River Basin.

The changes in pool frequency and quality and LWD levels can lead to shifts in species or age composition. Bisson and Sedell (1984) attributed a decline in coho and older age classes of cutthroat in western Washington to the loss of pool volume and LWD cover. They found that when an old-growth stream is clearcut there is a shift to higher percentages of 0+ steelhead and 0+ cutthroat trout and lower percentages of 0+ coho and 1+ and 2+ cutthroat (Bisson and Sedell, 1984). Riffles increased in length by a greater percentage and tended to eliminate many former pool areas. The increase in riffle area, decrease in pool volumes and loss of cover in pool areas favored a shift to those species and age classes most dependent on riffle habitats. Reeves et al. (1993) also found that salmonid diversity decreased with decreasing LWD and pool frequency in streams in coastal Oregon.

The success of in-channel approaches to habitat improvement by pool creation or LWD additions appears to have mixed results, although salmon response has not been well-documented. Frissell and Nawa (1992) documented that added structures (LWD and gabions) have limited effectiveness and longevity in streams with unstable banks, high flows, and high sediment loads.

They found little evidence that significant biological benefits have resulted from such structural measures. Platts and Nelson (1985) found that added LWD within a small exclosure had limited effectiveness in creating pools with continued heavy grazing upstream of the exclosure, although vegetation and channel form within the exclosure showed improvement. Platts and Nelson (1985) concluded, "...this improvement was countered by off-site, upstream influences and on-site, instream improvement structures that functioned as fine sediment traps. Fish populations did not respond... because the relatively small size of the livestock exclosure *did not reduce incoming, limiting influences created by upstream conditions*..." (emphasis added). Some evaluations have concluded that instream additions of LWD may harm streams that do not have adequate bank stability and riparian vegetation (Beschta et al., 1991). Instream efforts to form pools through addition of wood or other mechanical methods are unlikely to be successful if they do not address the cause of degradation (Beschta et al., 1991; Frissell and Nawa, 1992; Beschta et al., 1993; Kauffman et al., 1993) and may also damage streams by fixing their positions within floodplains and constraining their ability to form habitat through meandering (Beschta et al., 1991).

On Fish Creek in the western Cascades, it was not apparent whether massive increases in LWD and pools from in-channel restoration were successful at increasing the production of coho, chinook, and steelhead (Reeves et al., 1990). Some increase in coho production appeared to occur in an off-channel rearing pond (Reeves et al., 1990). The work of Reeves et al. (1990) appears to indicate that though pools and LWD increased dramatically, the biotic response may be indiscernible. A reason for this may be that other critical habitat variables remain limiting and that the full habitat improvement expected to stem from adequate LWD levels requires many years for adjustments in channel morphology to occur. The manual placement and cabling of LWD in selected locations, orientations, and volumes (primarily single logs rather than clusters) may never mimic the hydraulic role of natural LWD that can redistribute itself naturally.

In coastal Oregon, Nickelson et al. (1992b) reported positive responses in coho production to the creation of man-made alcoves to serve winter habitat needs, but similar success was not found in creating dammed pools. The addition of debris bundles (small trees) to dammed pools created by channel-spanning LWD was found to enhance winter rearing capacity for coho to levels comparable to those found in natural dammed pools (Nickelson et al., 1992b). Both natural alcoves and dammed pools (including beaver ponds) supported the highest winter densities of juvenile coho (Nickelson et al., 1992a) in areas where natural alcoves and beaver dams had been lost as a result of logging and agricultural activities (Sedell and Luchessa, 1982 as cited in Nickelson et al., 1992a). Creation of pool habitat was reported to increase summer rearing potential for coastal Oregon coho (House and Boehne, 1986). In summer habitat, Shirvell (1990) observed that coho, steelhead, and chinook all took positions in a British Columbia stream reach downstream from both artificial and natural rootwads. These structures provide the necessary conditions of low velocity flow and low light intensity. Rootwad introduction to a mid-channel area was found to improve the ability of fish to occupy the area.

The frequency and quality of pools and LWD strongly affects carrying capacity of available rearing habitat, so these attributes may, primarily, have a density-dependent effect on salmon populations. However, little is known about the effect of pool and cover loss on predation efficiency and some have argued that it is a density-independent effect (D. Chapman, Consulting Fish. Bio, pers. comm., 1992). Further, little is known about the synergistic effect on juvenile and adult salmon caused by the loss of cover and pools, combined with increased temperature extremes, reduced water flows, and increased fine sediment, all of which typically exist in areas where pool loss has been significant.

1.2.1.2 Activities Affecting Pools and LWD: Pool loss is caused by accelerated sediment delivery to streams (Lisle, 1982; Jackson and Beschta, 1984; Alexander and Hansen, 1986; MacDonald and Ritland, 1989; Lisle and Hilton, 1992; MacDonald et al., 1991; Chamberlin et al., 1991; Swanston, 1991; Platts, 1991; Boise National Forest, 1993) and/or the loss of LWD (Everest et al., 1985; Sullivan et al., 1987; Bisson et al., 1987; MacDonald et al., 1991; Hicks et al., 1991; Swanston, 1991; Nickelson et al., 1992a; Fausch and Northcote, 1992; Ralph et al., 1994). Many activities simultaneously increase sediment delivery while reducing LWD recruitment, stability and longevity.

Mining, logging, road construction, and grazing all increase sediment delivery at the watershed scale (See sections on channel substrate (1.1), riparian reserves (3.1), and sediment delivery (3.2)). However, these activities cause greater increases in sediment delivery with proximity to the stream system, with other factors kept equal. These activities can also reduce LWD levels over time through a variety of mechanisms when they occur within one tree height of floodplains.

LWD quantities are strongly controlled by the size and number of trees in riparian zones (Bisson et al., 1987; Sedell et al., 1988; Grette, 1985; Gregory and Ashkenas, 1990; McDade et al, 1990, Robison and Beschta, 1990). Tree size and stocking affect both LWD recruitment and stability. When riparian areas contain fewer and smaller trees, LWD levels decrease over time because recruitment generally occurs within one tree height of streams (Rainville et al., 1985; McDade et al., 1990; Robison and Beschta, 1990). Reductions in the size of LWD recruited also influences the longevity of LWD. Smaller pieces which have less longevity in the stream system because they are less stable within larger channels and decompose faster (Bisson et al., 1987; Sedell et al., 1988). Shifts in the composition of riparian tree communities can affect the recruitment and longevity of LWD. Coniferous LWD has greater longevity than LWD from deciduous vegetation due to its resistance to decay and greater size (Sedell et al., 1988; McDade et al., 1990). An ongoing supply of LWD is predicated on the continuous and adequate stocking of riparian areas with large trees. Over time, LWD decomposes or is removed by high flows (Bisson et al., 1987; Sedell et al., 1988). It is likely that LWD recruitment is reduced when trees are removed within a tree height of streams (Rainville et al., 1985; McDade, et al., 1990; Robison and Beschta, 1990). However, it also likely that logging within floodplains may also reduce LWD recruitment to streams over time as they migrate or during flooding. LWD loss can arise from active removal, loss of standing riparian trees and diminishment of recruitment potential, decomposition and transport from reaches which no longer have morphology or bank vegetation conducive to retention, stranding of small, de-stabilized LWD at channel margins and above high water, and reduction in input rates from upstream reaches.

Roads and mining within riparian areas severely disrupt LWD input because they prevent the regrowth of trees not only for the life of the activity, but for extended periods, because regrowth is slow on old roadbeds and mined areas. Timber harvest within riparian zones reduces LWD levels over time by direct removal of trees, altering community composition, and shifting trees towards smaller sizes. The loss of LWD sources is significant because the recruitment of coniferous LWD does not begin to occur again until about 50 years or more after removal (Bisson et al., 1987); more than 200 years is required for full recovery of LWD sizes and amounts in the western Cascades after logging (Gregory and Ashkenas, 1990). Due to slower rates of regrowth, the full ecological recovery time for LWD recruitment in the Snake River Basin may be even longer. Notably, the removal of riparian trees may exacerbate the loss of LWD not only by directly reducing recruitment, but also through the loss of bank stability leading to decreased longevity.

Many studies indicate that pool loss has occurred in response to increased sedimentation and reductions in the size, frequency, and stability of LWD. In California, streams estimated to have higher amounts of sediment delivery have been found to have a greater fraction of pools filled by fine sediment (Lisle and Hilton, 1992; Hagberg, 1993). Frissell (1992) found that pools were shallower in more heavily logged watersheds in coastal Oregon; LWD orientation in the more heavily logged watersheds appeared to be less conducive to pool formation, but high levels of sediment delivery were also implicated. Reeves et al. (1993) found that streams in watersheds that had more than 25% of the area logged had lower levels of LWD and pools than less heavily logged basins; the differences were statistically significant. In western Washington, unharvested and moderately harvested streams with gradients less than 3% had >50% pool area while intensively harvested sites with similar stream gradients had <40% pool area (Ralph et al., 1994. LWD was significantly smaller and pool depths were significantly shallower in intensively logged basins (Ralph et al., 1994).

In Salmon River tributaries in Idaho, Overton et al. (1993) found a statistically significant difference in pool frequency and LWD frequency between a watershed that had been logged, roaded, and grazed and a relatively undisturbed watershed. The relatively undisturbed watershed had about twice the LWD frequency and more than twice the frequency of pools greater than 3 ft in depth (Overton et al., 1993). Pocket pool depths were lower in the logged watershed (Overton et al., 1993).

Clearcutting of stream reaches reduces bank stability, decreases LWD recruitment, and increases sedimentation resulting in reduced pool area and volume. The LWD that remains in clearcut reaches is often ineffective at creating of habitat structure (Johnson et al., 1986; Hartman et al., 1987; Thedinga et al., 1989). Hartman et al. (1987) studied the effects of three different streamside treatments in alluvial floodplain areas of Carnation Creek, British Columbia: an intensive clearcutting in which riparian vegetation was felled across the stream and all merchantable volume was removed from the stream; a careful clearcutting to the stream margin in which trees were directionally felled away from the stream; and a leave strip of trees with a width varying from about 3 to 210 ft. In the intensive treatment areas, winter habitat was lost because the volume and stability of LWD decreased one year after logging, banks eroded, and maximum pool depth decreased. In the careful treatment area, LWD stability decreased, maximum pool depths decreased, and bank erosion increased. In the leave strip area the volume and stability of LWD remained constant but pool depth decreased because of deposition of fine sediments transported from upstream intensive sites. Hogan (1987) found that logging of watersheds in coastal British Columbia reduced the size and abundance of recruitable riparian trees and consequent changes in channel morphology. Loss of standing riparian trees led to a reduction in size of in-channel LWD. The smaller LWD of logged channels was predominantly oriented parallel to streambanks and did not effectively store sediment or scour pools. LWD was aggregated in large debris jams that stored large volumes of sediment in localized areas. Logged channels had reduced pool area and depth. Ralph et al. (1994) documented a similar sequence of logging effects on LWD and channel morphology. Hogan (1987) noted that unlogged channels had very complex morphology with frequent vertical steps and sediment storage areas. They also had greater variance in channel width. Complex stream margins have been found to create abundant juvenile fish rearing areas (Moore and Gregory, 1988). Burns (1972) found that stream sedimentation from logging along northern California streams caused loss of pool area, resulting in reduction of coho and large trout.

Grazing also contributes to pool loss by reducing bank stability, increasing sediment delivery, and shifting plant communities. Many lowland riparian zones that historically provided prolific salmon habitat are now nearly devoid of natural riparian plant communities. The lowland habitats of eastern Oregon presently support grass communities dominated by Kentucky bluegrass and cheatgrass. Historically, these zones supported densely rooted, diverse understories of native grasses such as tufted hairgrass, slender wheatgrass, red fescue, sedges, mannagrass, and numerous other species. These areas also often contained dense thickets of "cottonwood, birch, alder, willow, dogwood, hawthorn, currants, mockorange, snowberry, wild rose, and ninebark" (Soper, 1986) that provided woody debris, bank stability, overhanging cover, and diverse habitat for terrestrial insects that serve as fish food. Other riparian plant associations of lowland areas were dominated by tree species such as ponderosa pine or quaking aspen (Kovalchik, 1987). Long periods of intensive use of forage by livestock in ponderosa pine/common snowberry-floodplain associations, for example, has caused Kentucky bluegrass to become dominant and for soil compaction and continued grazing to prevent restocking of the coniferous overstory as it deteriorates. Continued overuse of aspen stands by livestock often results in their elimination and replacement by various forbs (Kovalchik, 1987). Although LWD from hardwoods has less longevity than coniferous LWD, it is an important source for pool formation and cover riparian complexes where conifers are absent. Grazing has probably reduced LWD in areas where it has reduced the stocking of deciduous trees, especially in areas once dominated by hardwoods. Grazing also reduces bank stability (Platts, 1991) which can contribute to pool loss.

Changes in channel morphology in unconstrained alluvial channels caused by the effects of land management can also alter pool frequency. For instance, primary pools are strongly associated with meander bends (Keller and Melhorn, 1973). Therefore, an increase in meander wavelength is likely to decrease pool frequency. In response to increased sediment delivery and streamflow, the meander wavelength of streams can increase (Schumm, 1969 (See Table 2)). Increased meander wavelength may also decrease pool depths at meander bends because pool depth tends to decrease with increased bend radius (Richards 1982); bend radius increases with increased meander wavelength. Mining, grazing, logging, and road construction all tend to increase sediment delivery and can increase streamflow (See sections on channel substrate (1.1) and sediment delivery (3.2)). However, alterations in channel morphology can be mediated by the condition of deep-rooted riparian vegetation along alluvial streams. Unconstrained streams lacking deep-rooted riparian vegetation are more sensitive to changes in channel morphology caused by changes in sediment load and discharge than well vegetated streams.

Channel widening can also accelerate LWD loss. It appears that there is a lower size threshold that is stable in channels of a given width (Bisson et al., 1987). As channel width is increased, smaller pieces of LWD may become unstable. Channel widening can occur in response to increased sediment loading (Howard, 1987) or the loss of bank stabilizing vegetation (Ikeda and Izumi, 1990; Platts, 1991). Grazed streams are often wider than ungrazed streams (Platts, 1991).

Pool loss has been tremendous over the past 50 years in most Snake River Basin streams in watersheds that have had significant amounts of mining, agriculture, grazing, road construction, and logging (Sedell and Everest, 1990; McIntosh, 1992; Boise National Forest, 1993; McIntosh et al., 1994 (See Figures 10, 11, 12, and 13)). In contrast, pool loss was insignificant in wilderness streams over the same time period (Sedell and Everest, 1990; B. McIntosh, USFS PNW Research Station Res. Asst., pers. comm., 1993). Streams with high levels of pool loss include the Lemhi, Stanley, Clearwater tributaries, Grande Ronde tributaries, and Middle Fork Salmon River tributaries (Sedell and Everest, 1990; McIntosh, 1992; B. McIntosh, USFS PNW Research Station Res. Asst., pers. comm., 1993; Boise National Forest, 1993). Low pool frequencies occur in many streams with high sediment loads and subjected to the combined impacts of mining, logging, and grazing (See Figures 14, 15, and 16). These low pool frequencies are probably the product of pool loss caused by high sediment loads and the loss of LWD. Available data indicate that pool conditions are poorest in watersheds with high sediment loads, grazing, and poor bank stability (See Figures 17 and 18). Available data indicates that pool frequency is inversely correlated with fine sediment levels ($p<0.10$) in watersheds in the Idaho batholith (See Figure 18 and Table 3). In the Upper Grande Ronde River Basin, large pools (depth > 3 ft) are lacking in subwatersheds with high sediment loads (See Figure 19).

At considerable time and expense, LWD has been added to many degraded streams throughout the Pacific Northwest to increase pool volumes. In the Snake River Basin, this effort has been largely unsuccessful because pool formation requires well-vegetated and stable channel banks that are lacking in most degraded streams (Beschta et al., 1991). In southwestern Oregon, LWD additions had limited longevity and effectiveness in forming pools in degraded streams with high sediment loads and unstable channel beds (Frissell and Nawa, 1992). The failure rate of in-channel structures was greatest in alluvial channels (Frissell and Nawa, 1992) that are typically the most productive salmon habitats. Platts and Nelson (1985) found that LWD additions did little to create pools when upstream sediment loads remained high. Quality pools are not typically created by engineered log steps (Nickelson et al., 1992b). Manual placement of LWD in stream channels does little to improve the natural LWD recruitment function of the riparian zone; in some instances, LWD additions have been made from standing riparian trees rather than from sources outside the riparian zone, reducing natural recruitment of LWD in the future. In many cases, logging, road construction, and mining have reduced LWD recruitment much more than can or has been added to a stream by enhancement projects.

Once pool volume has been lost due to sedimentation, the rate of recovery is partially dependent on the amount of on-going sediment delivered to the stream system (Platts and Nelson, 1985; Platts et al., 1989, Perkins, 1989; Bohn and Megahan, 1991). The recovery of channel morphology and pool volumes is slowed by high sediment loads. Efforts to rebuild pools through the addition of LWD are thwarted when upstream sediment delivery is not reduced (Platts and Nelson, 1985; Beschta et al., 1991). Pool excavation is also probably shortlived under high sediment loads because pools act as sediment traps and cobbles rapidly become embedded if levels of ambient fines are high (Reeves et al., 1991).

Reeves et al. (1991) concluded that it is far more cost effective to prevent habitat degradation than to try to restore it. Hicks et al. (1991) reviewed more than 30 years of research on forestry-fishery interactions and concluded that protection of intact riparian areas, floodplains, and side-channel habitats plus careful management of watershed sediment production are essential for maintenance of stream health.

1.2.1.3 Evaluation: Existing information amply indicates that LWD and pools are important to salmon survival and production. The best available information also indicates that there is a strong linkage to land use activities and resultant changes in LWD and pool conditions. It is also clear that the loss of pools and LWD is a problem in the Snake River Basin. Existing studies indicate that pool loss over the past 50 years in the Snake River basin has been significant (50-80%) in managed watersheds with highly elevated levels of sediment delivery and insignificant in wilderness watersheds (Sedell and Everest, 1990; McIntosh, 1992; B. McIntosh, USFS PNW Research Station Res. Asst., pers. comm., 1993; Boise National Forest, 1993; McIntosh et al., 1994). Pool loss was greatest in heavily grazed watersheds (B. McIntosh, USFS PNW Research Station Res. Asst., pers. comm., 1992; J. Sedell, USFS PNW Research Station Aquatic Ecologist, pers. comm., 1992). Low pool: riffle ratios are found in watersheds in the Idaho batholith with high levels of management-induced sediment delivery and low levels of LWD (Espinosa and Lee, 1991; Clearwater National Forest, 1991b).

Studies in western Washington and Oregon indicate that the size and frequency of LWD in streams has been radically altered by logging and road construction in riparian zones relative to that found in pristine systems (Everest et al., 1985; Bisson et al., 1987; Sedell et al., 1988; MacDonald and Ritland, 1989; Gregory and Ashkenas, 1990). It is highly likely that this is also the case in managed watersheds of the Snake River Basin given existing riparian conditions, logging levels, and their relation to LWD. In the Clearwater National Forest about 10 to 30% of the riparian areas in some logged watersheds have either been harvested within the past decade or roaded (Clearwater National Forest, 1991b and 1992). Many logged watersheds have low levels of LWD (See Figure 20) and future sources of LWD are limited (Wallowa-Whitman National Forest, 1992; Anderson et al., 1992). Due to the existing condition of riparian zones and sediment loads, it is likely that pool volumes and LWD levels will continue to deteriorate in all natal habitat in the Snake outside of wilderness/roadless areas, unless measures are taken to protect and re-establish LWD sources and bank stability and reduce sediment loads.

Many research studies have illustrated the importance of pool abundance and quality in determining the capability of a stream to produce fish (e.g., Nickelson et al., 1992a). However it is also likely that salmon production is also determined by the spatial organization, quality, and complexity of these units. For instance, a stream with alternating sequences of very long riffles and pools would likely have different fish production capability than a stream with sequences of short riffles and pools, all other factors being equal.

The different qualities of pools have led to a proliferation of habitat type descriptions using depth and/or flow velocity as criteria for delineation (Bisson et al., 1982; Platts et al., 1987; Hawkins et al., 1993). It has commonly been considered that the riffle/pool ratio (on the basis of area) and pool quality are a measure of the salmon production capability of streams (Platts et al. 1983).

There have been many recommendations for goals for pool frequencies for salmon streams. An optimum ratio of riffles:pools (based on area) is commonly taken as 50:50 for fish production (Armour et al. 1983; USFS, 1985; Reeves et al., 1989), although the validation of this concept is weak. Washington Forest Practices Board (1993) provides diagnostic indices of habitat quality for pool abundance and frequency for channels less than about 49 ft wide. Despite this, they caution against applying simple numerical pool frequency standards to all channel types. Their use of pool indices is primarily aimed at determining LWD needs based on the positive correlation between pool frequency and LWD volume. Eiserman et al. (1975, as cited in Armour et al., 1983) recommended that an optimum mixture of riffles, pools, and glides was 35:35:30. Bisson et al. (1982) studied four western Washington streams which were third to fifth order with gradient of 1-8% and had sites ranging from clearcut to second growth to old growth. They found that pools comprised approximately 50% of the stream by length and 80% by volume. Riffles accounted for 40% by length and 16% by volume. Bugert et al. (1992) reported that riffle:pool ratios changed substantially from one year to the next in a wilderness portion of the Tucannon River. Although many have focused on pool frequency, area, and/or volume, the spatial distribution of habitat types and riffle:pool ratios by reach in an entire drainage network probably interact in complex ways to generate a total basin's smolt production. Macrohabitat (channel units) such as riffle and pool types explains some of the distribution of fish biomass, but fuller descriptions of habitat at higher and lower spatial scales (e.g., see Frissell et al., 1986) are needed.

Proponents of the view that healthy stream reaches must have riffle:pool ratios of 50:50 and also a pool frequency of one per 5-7 times stream width must reconcile these two standards by distinguishing primary pools from secondary pools. The 50:50 ratio assessment is based on lumping pools and glides. This simplistic area ratio conceals important differences in slow water habitats. A reach comprised primarily of glides is biologically very different from one having numerous deep pools. Although trend monitoring data revealing widespread conversion of pool area to riffle is likely an important tool for indicating land management effects, the use of ratios of pools of various qualities, glides, and riffles as standards appears to be far less tractable as a general or even site-specific standard than other variables.

Pool and LWD metrics may not be desirable as screening standards for a few reasons. First and foremost, it is clearly desirable to manage streams for the highest natural rate of recruitment of LWD and the highest sustainable level of pool frequency, quality and volume, given their numerous ecological functions and direct importance to salmon survival and production. In the case of pools and LWD, there are land use standards that can serve as surrogates to in-channel standards that should adequately allow the protection and improvement of pools and LWD. Regardless of the current status of pools and LWD levels, the operant ecological processes and components that provide LWD and pools should be protected. These are identifiable and include the protection and re-establishment of bank stability, floodplain soils and vegetation, riparian forest communities, and nearly natural sediment delivery regimes.

Second, both pools and LWD are expected to be somewhat temporally insensitive to changes in land management. Therefore, monitoring a standard specifying maintenance of the abundance or frequency of pools or LWD is fraught with problems of detecting change. For instance, the average rate of pool loss found by Sedell and Everest (1990) is about 50% loss in 50 years; if one assumes that initial pool frequency was one pool every 7 channel widths and channel width is 30 feet, there would have been 100 pools over an instream distance of 4.4 miles. Even though a 50% pool loss in 50 years is very significant, detection requires discerning an average yearly loss of 1 pool per 4.4 mile distance. Under non-catastrophic conditions, pool loss may occur gradually through pool in-filling and reductions in depth. Thus, the high rate of pool loss documented by Sedell and Everest (1990) might be difficult to detect except after long periods if pool loss proceeds uniformly in time. Ralph et al. (1994) determined an effect of channel gradient and logging intensity on pool frequency and area in low gradient streams but concluded that riffle:pool ratio is not useful as a measure of land use because it is not repeatable and is subject to pool identification problems. These pool indices were not as useful for detecting intensive land use effects in high gradient streams (>8%) as in low gradient streams (Ralph et al., 1994). In some stream systems, pool volume and depth appear to be more sensitive indicators of the effects of land management on habitat conditions than pool frequency (Frissell, 1992; Ralph et al., 1994).

The catastrophic sedimentation of the South Fork Salmon River demonstrated that pool loss can happen abruptly. Although the monitoring of pool frequencies can easily detect such radical shifts, there is little that can be done to rapidly reverse such damage.

Although assessments of pool volume and frequency are important diagnostic tools for habitat degradation, recovery, and suitability for salmon production, we do not recommend that pool frequencies be used as a standard in the process of coarse screening for assessing the consistency of management actions with protecting and improving salmon habitat. A better approach to providing adequate pool frequencies and volumes is to establish riparian reserves that fully protect floodplains and LWD sources, meet bank stability standards and limit sediment delivery (See sections on channel substrate (1.1), bank stability (1.2.2), riparian reserves (3.1), and sediment delivery (3.3). Over time, pool area and frequency should reach equilibrium with reduced sediment delivery and increased LWD recruitment. This is virtually the only way to reestablish desirable primary pool frequencies and diversity of pool qualities in a sustainable fashion. Despite this, the key biological significance of pools in salmonid production makes pool quantity, frequency, and quality variables worth monitoring. If improving trends in these pool indices do not occur while implementing all other recommended restoration actions, further restoration efforts should occur, including continued reduction in sediment delivery and/or restoration of sources of natural recruitment of LWD.

We recommend that residual pool volume be monitored using the methods of Lisle and Hilton (1992) be initiated to establish trends in available pool volumes. Residual pool volume is the volume of a pool not filled by fine sediment accumulations up to a pool surface elevation equivalent to the pool tail bed elevation (control point). Fine sediment volumes in pools reduce pool quality and reduce residual pool volumes (the minimum pool volume available for salmon use). We cannot recommend a numeric standard for residual pool volumes because linkages between residual pool volumes and fish production have not been well-established. However, we do recommend that watersheds should be managed so that there is a decrease in fine sediment volumes in pools and increased residual pool volumes in managed watersheds. These pool variables hold considerable promise as "early warning indicators" of trends in pool frequency and volume. Where there is an increasing trend in fine sediment volumes in pools, sediment delivery should be reduced through passive and/or active watershed restoration.

We recommend against the construction of pools in channel segments. It is not desirable to attempt to force a stream to adopt a channel morphology that is not physically sustainable. The guiding principle that should be followed in stream restoration work is to allow the stream to express its dynamic channel morphology by fully protecting or recovering floodplain function and by reducing anthropogenic sources of watershed disturbance. Full protection of floodplain, riparian, and channel function throughout a stream network should allow development of appropriate riffle:pool ratios, pool frequencies, and habitat qualities.

Numeric standards for in-channel LWD frequency have been used by some entities to set riparian width and tree stocking density assumed to be adequate to provide these levels of LWD. This approach generally has resulted in programmed degradation of salmon habitat. Given the available information, any reduction in stable LWD acts to reduce the quality of salmon habitat and has the potential to reduce pool volumes. Among the other flaws inherent in meeting numerical LWD targets are the following: (1) The LWD standards are frequently based on surveys of streams that have been impacted by riparian harvest; (2) Minimum LWD levels are typically selected because it is assumed that any reach that could not naturally be expected to meet the standard would invalidate the standard; (3) Deciduous leave trees are allowed to replace harvested coniferous trees in mixed stands; (4) Minimum diameters and lengths of LWD are taken as standards; (5) It is assumed that geomorphically stable riparian areas do not need tree cover because root protection of banks is not important; local biological needs for LWD (fish habitat, water temperature control) are not significant, and flotation of LWD to downstream reaches does not occur; (6) Past site conversions from forest to pasture or road surface are considered to be irreversible past land use decisions; (7) Lowland streams in grazed lands typically have no LWD requirements because of faulty assumptions about riparian vegetative potential and substitution of forage standards for plant community standards; (8) The role of LWD in headwall stability is ignored; (9) It is frequently assumed that, even though very large LWD dominates low-order old growth streams, this material can be functionally replaced by small LWD and younger riparian stands. It is typically assumed that because one of the key roles of LWD is to create pools, a pool engineered with one log of any size is equal to one formed by complex, natural accumulations of LWD.

Rather than specify numeric standards for LWD, we recommend full protection of riparian systems and establishment of riparian reserves as a surrogate approach. This recommendation is made for two reasons: 1) the recommended riparian reserves should fully protect existing LWD sources and allow re-establishment of LWD recruitment; 2) trends in LWD will probably not be rapidly established and are not amenable to a timely monitoring/management feedback approach. LWD loss can be gradual even after recruitment sources have been completely truncated by the clearcutting of reaches (Grette, 1985). It is much more efficient to protect sources of LWD rather than waiting to detect losses below a management threshold.

In many watersheds, it is highly probable that establishment of riparian reserves will ultimately result in the adequate recruitment of LWD via the protection of continued and sustainable sources of recruitment. This may not be the case in watersheds where riparian zones have been heavily roaded and/or mined; in these cases, active restoration approaches aimed at re-establishing tree stocking should occur. Although we do not recommend setting LWD levels as a standard in the screening process, we do recommend that LWD be monitored for trends.

We recommend that LWD additions aimed at forming pools should be undertaken only where: 1) it is ecologically appropriate given the types and existing conditions of the streams and riparian vegetation; 2) the causes of pool loss and channel instability have been adequately addressed, e.g., loss of LWD sources, sediment loading, bank instability, etc; and 3) it is likely that pool and LWD frequency is a major limiting factor to salmon production (e.g., water temperatures and other factors are amenable to salmon). Under no conditions should LWD additions be considered a surrogate for the protection or restoration of natural LWD sources.

1.2.2 Bank Stability

1.2.2.1 Effects on salmon: Bank stability is of prime importance in maintaining habitat conditions favoring salmon survival. Loss of bank stability increases erosion and sediment delivery (Richards, 1982). Increases in erosion and sediment delivery lead to channel widening, pool loss, and increased fine sediment and cobble embeddedness. Although other factors contributed to substrate conditions in Bear Valley Creek in Idaho, the Boise National Forest (1990) found that fine sediment levels were inversely related to bank stability. The inverse relationship between bank stability and fine sediment levels appears to generally hold among watersheds in the Idaho batholith based on limited data (R^2=0.46, $p<0.10$ (See Figure 6 and Table 3)).

Channel widening can de-stabilize smaller pieces of LWD that are no longer stable at new channel widths (Bisson et al., 1987). Channel widening also exacerbates seasonal temperature extremes. Channel widening reduces the effectiveness of shading by vegetation of a given height as well as increasing the net heat load stream under a fixed heat budget (Theurer et al., 1984; MacDonald et al., 1991). During winter periods, increased channel width increases heat loss and renders streams more susceptible to icing. Anchor ice can cause complete mortality of most aquatic life within the stream substrate (Platts, 1984).

Loss of bank stability can also result in stream incisement. Stream incisement can desiccate riparian areas causing a shift in plant communities towards more xeric assemblages. Desiccation of riparian areas due to stream incisement may also aggravate seasonal water flow extremes because groundwater inflows can mediate stream temperatures. Stream incisement may also reduce stream discharge during baseflow.

1.2.2.2 Activities affecting bank stability: Activities that increase peak discharge, reduce streambank vegetation, and/or increase sediment delivery can reduce bank stability. At the watershed scale, logging, mining, grazing, and roads can increase peakflows, especially in snow dominated climates, through a variety of mechanisms including increased snow accumulation and snowmelt rates in created openings, soil compaction, and decreased evapotranspiration caused by vegetation loss. Mining, roads, logging, and grazing near streams can remove vegetation that provides bank stability from root strength (Graf, 1979; Richards, 1982). Grazing also reduces bank stability via the trampling and chiseling of channel banks as documented by field reviews and monitoring (Beschta et al., 1991; J. Rhodes, unpublished field notes, 1991 and 1993; Platts, 1991; Boise National Forest, 1993; Beschta et al., 1993). Although some monitoring has indicated that bank stability is inversely correlated to forage utilization (Boise National Forest, 1993), field evaluations of conditions at the end of the grazing season in some wet meadow systems in the Blue Mountains of Oregon have shown extreme losses of bank stability that were not related to forage utilization, but caused by trailing and trampling along channels (J. Rhodes, unpublished field notes, 1993). Grazing is typically cited as having the greatest effect of land management activities on bank stability (Platts, 1991; MacDonald et al., 1991); however, mining close to streams can have devastating effects on bank stability (Nelson et al., 1991). Field reviews have noted that stream incisement caused by grazing is fairly common in the grazed watersheds in the Snake River Basin and in the Blue Mountain Province of Oregon (Beschta et al., 1991; J. Rhodes, unpublished field notes, 1989; 1990; 1991; 1993; Platts, 1991; Rhodes et al., 1993). Bank stability and substrate conditions are generally poor in most grazed watersheds in the Snake River Basin, but are especially poor in watersheds that have also been roaded, mined, and logged (See Figures 21 and 22).

1.2.2.3 Evaluation: It is clear that low bank stability is a common problem in Snake River Basin habitat and that it contributes to sediment loads, poor channel morphology, and high summer water temperatures.

We recommend that watersheds should be managed so that more than 90% of channel banks on all streams are stable. Where bank stability exceeds 90%, watersheds should be managed so that there is no decrease in bank stability. In watersheds where bank stability is less than 90%, or there is a decreasing trend in bank stability ($p<0.40$), activities that can potentially decrease bank stability or forestall recovery should be eliminated until the standard has been reached or a statistically significant ($p<0.05$) improving trend over at least five years has been documented through monitoring. Suspension of riparian grazing in degraded reaches is one of the key strategies to restoring bank stability. We recommend that grazing be suspended within one- half tree height of floodplains or streams, whichever is greater, in all reaches or watersheds where the bank stability standard is not met until the standard is met or a statistically significant ($p<0.05$) improving trend over at least five years is documented through monitoring. Suspension of grazing within these distances of the streams should allow the recovery of streamside vegetation over time that will aid in stabilizing stream banks in the long term. Although only one-half tree height from the edge of a stream is all that may be needed to restore bank stability for the stream's current position (USFS et al., 1993), it is prudent to attempt to restore bank-stabilizing vegetation throughout the floodplain to provide stream protection as streams meander naturally across floodplains.

We recommend that efforts to stabilize banks should focus on protecting and restoring riparian vegetation. We strongly recommend against mechanical channel stabilization methods. These approaches can shift bank instability problems downstream and tend to fix channels to positions within floodplains, thwarting the ability of a stream to create complex habitat features, such as side channels and meander bend pools. Mechanical bank stabilization approaches are ecologically unsound and can create more and worse problems than they aimed at solving. We recommend that the primary approaches to protecting and restoring bank stability should be to
fully protect floodplain vegetation, soils, and banks from the physical insults caused by livestock trampling, road crossings, and other engineered, structural or mechanical perturbations.

1.3 WATER QUALITY

1.3.1 Water Temperature

1.3.1.1 Effects on salmon: The effects of water temperature on salmon are probably the most clearly understood and extensively documented of any habitat factor affecting salmon survival and production. They are also the most amenable to quantitative prediction (Armour, 1991).

Water temperatures in excess of the optimum range in rearing habitat lead to decreased growth and increased mortality in salmon in a variety of ways, including: 1) reduced ability to compete for food and avoid predators due to physiological stress; 2) decreased dissolved oxygen capacity of water; 3) increased oxygen demand due to increases in respiration; 4) increased competition from warm-water fish for food and space; 5) increased incidence and virulence of diseases affecting salmon; and 6) the increased toxicity of many substances; and 7) reduced usable rearing area (Everest et al., 1985; Theurer et al., 1985; Beschta et al., 1987; Armour, 1991). For these reasons, some researchers have considered an average daily temperature of 68°F to be the upper lethal limit due to these combined indirect effects (Theurer et al., 1984; Theurer et al., 1985). It is plain that the production of anadromous fish is progressively reduced at water temperatures in excess of 59°F (Everest et al., 1985; Armour, 1991). In addition, as food availability decreases, the optimum temperature for salmonid growth declines. Because of this effect, the impact of increasing temperatures may be greater than predicted in the field if food supplies become limiting (Elliott, 1981).

Water temperatures of approximately 77°F cause direct mortality in spring chinook salmon (Bell, 1984; Theurer et al., 1984). Bouck et al. (1970, as cited in USEPA et al., 1971) determined that temperatures >68°F were not safe for adult sockeye because of increased mortality from disease. A study by Fish (1944, as cited by Parker and Krenkel, 1969) revealed that adult sockeye mortality was 49% when adults were subjected to fluctuating temperatures of 49-74°F (**b**=61°F) but were only

4% at 52-60°F (**b**=53°F). The upper limits of water temperatures for the successful migration of returning spring and summer chinook are about 56°F and 68°F, respectively (Olson and Foster, 1955; Bell, 1984; Bjornn and Reiser, 1991; Armour, 1991). Acceptable fall chinook migration temperatures (incorporating a more limited optimal range) are 51-67°F for the Pacific coastal region (Bell, 1984). Temperatures >70°F created a migration blockage for sockeye migrating up the Okanogan River from the Columbia River (Major and Mighell, 1966, as cited by Parker and Krenkel, 1969). Hatch et al. (1993) reported that as water temperature reached 73°F in the Okanogan River, sockeye passage upstream to Lake Osoyoos terminated. During the migratory period, sockeye did not migrate from the Columbia River staging area upstream on the Okanogan River until temperatures dipped below 73°F. Bell (1984) reports 70°F to be a temperature at which sockeye migration is delayed. Acceptable migration temperatures range from 45 to 60°F. Even though Hatch et al. (1993) observed
initiation of migration below 73°F, it appears likely that the optimal migration temperature is about 58°F, the temperature at which the sockeye cruising speed is maximum (See Bell, 1984).

Temperatures higher than optimal can contribute to elevated levels of pre-spawning mortality in returning adult salmon, although it has not been well-documented in the field. The pre-spawning suitability index for chinook is maximal between 46-55°F and is zero at >75°F (Raleigh et al., 1986). Exposure of pre-spawning adults to temperatures of 60.8°F was reported to result in 64% mortality in 56 days (Reingold, 1968, as cited in Parker and Krenkel, 1969). Temperatures below 60°F reduce the probability of infectious warmwater diseases in adult chinook (USEPA et al., 1971). Temperatures of 68°F have resulted in 100% mortality of chinook during columnaris outbreaks (Ordal and Pacha, 1963). For adult sockeye on Fraser River spawning grounds, mortality of females ranged from 5-86% from gill bacterial infections at temperatures of 72°F (International Pacific Salmon Fisheries Commission, 1962, as cited by Parker and Krenkel, 1969). Columnaris in sockeye becomes increasingly active above 60°F (Colgrove and Wood, 1966, as cited by Parker and Krenkel, 1969) and has been implicated in high sockeye mortalities in the Columbia River (Fish, 1948, as cited by Parker and Krenkel, 1969). Bouck et al. (1970, as cited by USEPA et al., 1971) did not observe sockeye mortalities from columnaris at 62°F but did note frequent lesions and death at 68°F.

In addition to direct effects on adults, egg survival is only 70% when pre-spawning adults are subjected to water temperatures of 60-62°F and eggs are then incubated at 55-56°F (Hinze, 1959, as cited by California Department of Water Resources, 1988). Water temperature of 58°F is the upper limit of the optimum water temperature range for incubation of salmon eggs (Bell, 1984; Bjornn and Reiser, 1991). Salmon eggs incubated in water temperatures in excess of 58°F incur increased mortality (McKee and Wolf, 1963).

Incubating eggs and alevins are also affected by low winter temperatures. Bjornn and Reiser (1991) recommend incubation temperatures of 41.0-57.9°F. When temperatures decline to less than 34-35°F survival decreases to 0% (Seymour, 1956, as cited by California Department of Water Resources, 1988; Combs, 1965, as cited by Parker and Krenkel, 1969). The development of anchor ice smothers eggs and can result in the complete mortality of eggs and alevins in the stream substrate by suffocation or physical dislodgement (Platts, 1984; Everest et al., 1985).

A water temperature of 59°F is considered optimum for rearing chinook (Armour, 1991). Temperatures above 59°F and below incipient lethal levels are tolerable but increasingly impair fish growth and survival (Theurer et al., 1984; Everest et al., 1985; Theurer et al., 1985, Armour, 1991). Bjornn and Reiser (1991) list temperature preferenda for chinook and sockeye juvenile rearing as 54-57°F. The water temperature preferenda for spring chinook by lifestage are diagramed in Figure 23. The preferenda found in Figure 23 are based on values developed from a comprehensive literature review (McCullough, *in process*).

Thermal refuges, where available, can serve an important role in increasing salmon survival during winter as well as summer. Groundwater entry in stream channels provides important overwintering areas for salmonids owing to the more stable flows, thermal regimes, and macrophyte hiding cover (Cunjak and Power, 1986).

Alteration of the water temperature regime has both density-dependent and density-independent effects on salmon populations. Density-dependent effects of temperature on fish under thermal stress within the tolerance zone (See Elliott, 1981) in the field may be heightened by temperature avoidance and crowding into remaining marginal habitats. As temperatures increase further and enter the lethal zone, either the coldwater population continues avoidance behavior and becomes more crowded in refuges, or density-independent mortality exerts a stronger influence. Water temperatures in the lethal range will produce density-independent mortalities no matter how high habitat quality is in a particular stream reach. If habitat quantity in thermal refuges is not sufficient, density-dependence will limit the extent of crowding. Factors such as heightened disease contagion under crowded conditions in adverse water temperatures can also increase mortality.

Elevated water temperatures and water diversions have reduced usable rearing habitat in many natal streams. If rearing juveniles can actively avoid adverse temperatures and/or dewatered reaches and crowd into suitable habitats (limited thermal refuges, coldwater tributaries, headwater areas) density-dependent population controls can become prominent, as previously stated. Shifts in competitive advantage for food and space requirements among warmwater tolerant and intolerant species or increased predation by warmwater species on coldwater species under general increases in water temperatures adversely affect coldwater species by reducing growth rate, survival, and spatial distribution. The magnitude of effects of competition and predation is relative to the population densities of warmwater tolerant vs. intolerant components of the community. The low population densities of many salmonid species make these temperature-mediated biological interactions a serious threat. Mortality during streambed icing conditions is largely density-independent. However, because the impact is spatially heterogeneous, affecting shallower, slower moving stream zones to a greater extent, the percent mortality to a population might be increasingly severe at larger overwintering population sizes provided that a large fraction of the population is forced to inhabit less favorable, shallower sites.

Effects of elevated temperatures on rearing juveniles or pre-spawning adults in the tolerance zone where loading stresses accumulate (Elliott, 1981) are primarily a function of temperature on fish physiology. In this zone, lowered resistance to disease, increased rate of death after exposure to disease, reduced growth rate, increased respiration rate, reduced swimming capability and feeding response, and increased stress response all contribute to increased probability of death. Likewise, mortality at elevated temperatures within the incipient lethal range is primarily related to exposure and acclimation times, but is also partially related to the prior condition of fish going into the lethal temperature zone (zone of resistance) (see Elliott, 1981). Although the beneficial effects of thermal refuges in maintaining populations under adverse environmental conditions are often emphasized, many years of stream channel modification and the loss of LWD, wetlands, and pools have resulted in severe cumulative habitat degradation, limiting the distribution, size and quality of thermal refuges. The multiple biological (physiological, behavioral, ecological, pathological) effects of elevated water temperatures on juvenile salmon combined with the synergistic effects of other forms of habitat degradation accompanying temperature increases make it likely that salmon survival in the field under specified temperature conditions is far less than that indicated by short-term laboratory experiments.

Laboratory evidence for optimal salmon rearing temperatures can be contrasted with field evidence. Most laboratory studies have been conducted under conditions of constant acclimation and exposure temperatures with most other stress factors (e.g., competition) being eliminated. Disease normally does not affect test results significantly because of the short duration of laboratory experiments. Under field conditions, fish rear in fluctuating temperature conditions and can have options for dealing with elevated temperatures (entering the substrate where temperatures may be slightly colder, seeking cold groundwater seeps arising from streambeds or in deep pools, migrating upstream to seek temperatures within the preference range). Under field conditions, fish are also exposed to other stress including the cumulative effects of temperature exposure, other habitat factors, and associated biotic effects (disease, competition, predation). Consequently, the mean or maximum daily temperatures in which salmon are observed in the field provide indications about their preference and resultant biotic interactions; the temperatures where they are not found indicate temperatures they do not prefer, temperatures that may be lethal over extended exposures, and/or effects of biotic interactions.

Lindsay et al. (1986) presented clear evidence regarding the effects of maximum water temperatures on spring chinook distribution. They found that distribution of spring chinook fingerlings after emergence in the John Day River, Oregon extends downstream from the three primary spawning areas. From the spawning areas in the North Fork, fingerlings extend their distribution downstream below the North Fork mouth. As water temperatures increase in early summer, juveniles migrate back upstream. On the North Fork, the lower limit of juvenile rearing retreated above river mile 70 in response to increased temperatures (Lindsay et al., 1986 (See Figure 24)). A similar pattern of juvenile movement downstream after emergence, followed by a return movement upstream during July-August in response to increasing temperatures was observed in the Middle Fork. Temperature data from thermographs in the North Fork and Middle Forks of the John Day River allow regressions to be developed expressing the downstream boundary of distribution in relation to the water temperature at the thermograph.

The regression for the North Fork of the John Day River indicates that when the mean maximum water temperature was 73°F for a two-week period at a point location in the North Fork (i.e., at the recorder, river mile 44), no juveniles reared below that point (See Figure 24). A similar analysis of the Middle Fork data revealed that when mean maximum water temperature was 67°F at a point on the mainstem for a two-week period prior to sampling, no juveniles were found below this point (see Lindsay et al., 1986). This study clearly shows that available rearing area decreases as water temperature increases.

Data from Bugert et al. (1992) on the Tucannon River, in southeastern Washington can be used to infer summertime temperature limitations on spring chinook rearing distribution. In July-August 1990 they surveyed spring chinook parr densities in the lower 25 mile river section and found no parr at all. In 1990 and 1991 only 2 redds each were observed in this section; no redds were observed in 1986. Although water temperature data are lacking for this period in 1990, August temperature data for 1991 in this reach indicate that daily maxima of 81°F were observed. The mean daily temperature for all 31 days of August 1991 was 72°F; the mean maximum temperature for this same period was 77°F. Theurer et al. (1985) estimated that no spring chinook production would occur on sections of the Tucannon River where mean daily water temperature for July exceeds 68°F and the average maximum daily July water temperature exceeds 75°F. Consequently, they estimated that about 24 miles of the Tucannon mainstem had been lost as usable habitat due to increases in summer water temperatures (See Figure 25); they estimated that the elevation of water temperature had reduced production capacity from 2,200 to about 900 adult spring chinook salmon (See Figure 26).

Laboratory data on survival at constant temperatures with various acclimation temperatures can be used to predict survival in the laboratory under fluctuating temperature regimes and thereby infer survival in the field. For example, considering the August 1-9, 1991 period in the lower Tucannon River (below river mile 25), one can calculate approximate hours of each day spent at temperatures above 75°F. Using coefficients for 90% mortality in thermal exposure tests for Columbia River spring chinook with 68°F acclimation followed by exposure to temperatures between 75 and 82°F (Blahm and McConnell, 1970, as cited in National Academy of Sciences, 1973), times to mortality can be calculated. Given the daily temperature cycles in this 9-day period, the daily cumulative percentage of a lethal dose was calculated for temperatures fluctuating in this range, using methods of DeHart (1975) and Golden (1978) (McCullough, *in process*). One can calculate by this procedure that chinook that may have been rearing in the lower Tucannon River would be exposed to 81.5% of a lethal thermal dose for a single day's temperature cycle. Combinations of 2 or 3 days exposure to these cycles would result in a full lethal dose.

Potential juvenile salmon growth rates under fluctuating summer temperatures can be associated with field distributional limits. The growth curve developed by Brett et al. (1982, as cited in Armour, 1991) indicates that growth rate at 66°F is about 60% of the optimum rate occurring at 59°F, while zero growth occurs at about 70°F for spring chinook. However, Hokanson and Biesinger (unpublished report, cited by Armour, 1991) indicate that 66°F is the temperature at which growth becomes zero for spring chinook. If 66°F is the true zero growth threshold (i.e., the temperature above which growth becomes negative), one can calculate from temperature data of Bugert et al. (1992) that during August in the lower Tucannon (up to river mile 26), the growth rate of rearing salmon would have been negative about 97% of the time. The extensive period of adverse juvenile growth conditions and estimations of time to achieve a lethal thermal dose in this lower river section both indicate that typical water temperatures in the lower Tucannon in August are intolerable to rearing salmon (Bugert et al. 1992).

Temperature data in August 1991 for monitoring sites at river mile 36 and 39 on the Tucannon reveal that temperatures exceeded 66°F for 58 and 63% of the month respectively. The site at river mile 39 had a 6-consecutive-day period with daily temperatures above 66°F for >80% of the day; the site at river mile 36 had a 5-day period with temperatures above the threshold for >90% of each day. Field surveys by Bugert et al. (1992) indicated that parr were found upstream of river mile 26 during the extended sampling period of August and September. However, because parr counts in August were not differentiated those in September, it is possible that low rearing densities in August were enhanced in September if downstream migration occurred during this period as temperatures declined. Equally possible is that parr survival in these warm river reaches was facilitated by localized pockets of cooler water derived from groundwater entry. However, with so much time spent above the growth threshold in August, it appears that a significant portion of the Tucannon River (i.e., from the mouth upstream at least to river mile 39) provided very low growth potential in August.

Field data on the distribution of resident salmonids in response to water temperatures provides some additional evidence of the effect of water temperature on salmon because there salmonids have exhibited broadly similar in responses to elevated water temperature in laboratory experiments. For example, a wide variety of salmonid species have ultimate upper lethal temperature levels in the range of 73-78°F. This range accounts for the response of the species chinook, coho, sockeye, chum, and pink salmon, steelhead, Atlantic salmon, brown, brook, and lake trout (Brett, 1952; Blahm and McConnell, 1970; Bishai, 1960; Fry et al., 1946; Fry and Gibson, 1953; Coutant, 1970, all as cited in National Academy of Sciences, 1973; Hokanson et al., 1977). The preferred temperature ranges for juvenile chinook, coho, sockeye, chum and steelhead are very similar and span 50-57°F (Bjornn and Reiser, 1991). Bull trout, however, have requirements for considerably colder rearing conditions.

Relatively abundant information from field studies exists on the effects of temperature on the distribution of salmonids at high temperatures. Dimick and Merryfield (1945) reported that no salmonids occurred in the Willamette River system where water temperatures exceeded 73°F; the majority of salmonids were associated with water temperatures ranging from 55 to 66°F and were always in lower abundance within the temperature range of 67 to 72°F. Upper temperature distributional limits for trout in southern Ontario streams have been reported as 75-78°F (Ricker, 1934; Barton et al., 1985). Such observations imply a continual decrease in salmonid density as temperatures increase from 60°F to >73°F. Similarly, Li et al. (1992) reported a decline in steelhead biomass from 0.37 lb/100 ft^2 at a maximum summer water temperature of 60.8°F in tributaries of the John Day River to 0 lb/100 ft^2 at a maximum temperature of 82.4°F. Hokanson et al. (1977) reported that rainbow trout reared under a fluctuating temperature regime of 71.6 ± 6.8°F had a specific growth rate of zero and a mortality rate of 42.8% per day, whereas mortality for trout held under optimal temperatures was only 0.36% per day. They estimated that even a mean weekly temperature of 62.6 ± 3.6°F for rainbow trout experiencing fluctuating temperatures in the field would result in a 27% reduction in production of the population over that under optimal conditions.

The sharp reduction in steelhead biomass with increasing temperature found by Li et al. (1992) is an indication of either progressive mortality or emigration from zones exceeding temperature preferenda. The ability of steelhead in the John Day River tributaries to tolerate such temperature extremes may be due to the availability of cooler water temperature refuges. These areas would be expected to severely contract in proportion of the wetted stream area with reductions in the volume of deep pools and increase in ambient water temperatures. Crowding of rearing juveniles into a decreasing pool area increases competition for food and space and results in density-dependent control on fish survival. Reeves et al. (1987) found steelhead to be dominant in steelhead/shiner interactions in laboratory streams when water temperatures ranged from 54-59°F but that shiners were dominant when water temperatures were 66-72°F. This study indicates that in addition to the lethal effects of temperature that become prominent above 72°F, negative competitive interactions reduce the ability of salmonids to maintain feeding stations and grow in streams with temperatures above this threshold. Ricker (1934) suggests that in addition to effects of increasing temperature, reduction in trout abundance is hastened by presence of warmwater tolerant fishes. Those salmonids attempting to inhabit the warmer stream zones have higher probabilities of dying from loading stresses, including disease and negative growth. When fish spend significant proportions of a day at temperatures in the "resistance" zone (See Elliott, 1981), lethal temperature doses are more clearly a function of exposure time to high temperatures and acclimation temperature.

Field studies reviewed for chinook, steelhead, and rainbow trout indicate that the distributional limit of these salmonids corresponds approximately to a mean daily water temperature of 68°F and a maximum daily water temperature of 73-75°F based on a 2-week exposure. This distributional limit, however, is a point at which biomass decreases to zero. Conditions for growth and survival immediately upstream from this boundary are apt to be marginal for growth and survival. Chinook growth in laboratory studies was reported as zero when water temperature is 66°F (See Armour, 1991). Laboratory studies on a wide variety of salmonids have indicated an ultimate upper incipient lethal limit of 73-78°F. The similarity in critical temperature indices from field and laboratory research suggests that laboratory studies on growth and survival at elevated temperatures can be used to explain distribution of the species in the field. However, as Hokanson et al. (1977) pointed out, even if a species such as chinook has zero growth at a mean daily temperature of 66°F under a fluctuating temperature regime, the population production would be negative because of the mortality rate. Because many prominent salmon diseases become virulent above 60°F, the impact to population production becomes more severe as temperatures rise.

Evidence reviewed on the water temperature thresholds limiting the distribution of various salmonids in the field indicates that temperatures exceeding a maximum of about 73°F results in very low to zero densities. Temperatures between the growth threshold (66°F) and 73°F will result in periods of negative growth for spring chinook juveniles (unless time spent below 66°F can compensate for time spent above this threshold) as respiration exceeds growth. Because growth rates decline from the growth maximum at 59°F to zero growth at 66°F, it is likely that rearing conditions become increasing marginal as temperatures are elevated to 66°F and beyond. The warmwater disease threshold is exceeded beyond about 60°F. At this temperature diseases such as columnaris and furunculosis become extremely active and deadly. Adverse interspecific competition becomes intensified between 59 to 66°F (See Reeves et al., 1987), at which point salmonids could be expected to be at a disadvantage relative to warmwater tolerant species.

1.3.1.2 Activities affecting water temperatures: The relationship between water temperature and the factors controlling it are well-understood and amenable to quantitative prediction. Land use activities that affect discharge, channel morphology, and streamside vegetation cover profoundly affect water temperature. Other factors remaining equal, streams with lower discharge are more susceptible to temperature increases during the summer and decreases during winter (USFS, 1980; Theurer et al., 1984; Everest et al., 1985; Beschta et al., 1987; Chamberlin et al., 1991). Likewise, shallower and wider streams are more susceptible to the development of seasonal temperature extremes (USFS, 1980; Theurer et al., 1984; Everest et al., 1985; Beschta et al., 1987; Platts, 1991).

Vegetation exerts strong control on stream temperatures; any loss of stream shading can increase summer water temperature, at least incrementally (USFS, 1980; Theurer et al., 1984). The removal of streamside vegetation is the dominant cause of elevated water temperatures in the northwestern U.S. (Everest et al., 1985; Beschta et al., 1987). Vegetation removal also increases the susceptibility of streams to icing (Platts, 1984; Everest et al., 1985; MacDonald et al., 1991; Platts, 1991). Small clearcuts increase water temperatures by 1-5°F (Beschta et al., 1987). Grazed streams typically have considerably less shade, higher water temperatures, and wider channels than ungrazed streams (Platts, 1991). Even very light grazing retards the regrowth of shade-providing vegetation (Green, 1991; J. Kauffman, Ore. State Univ. Prof. of Rangeland Resources, pers. comm., 1992). Although some grazing strategies are believed to allow the recovery of vegetative shading, vegetative recovery of shading is most rapid when grazing is eliminated from riparian areas (Platts, 1991; Green, 1991; Elmore, 1992; J. Kauffman, Ore. State Univ. Prof. of Rangeland Resources, pers. comm., 1992). Riparian zone recovery is not possible without at least temporary reductions in the number of livestock and grazing season (Ohmart and Andersen, 1986). Grazing can also exacerbate temperature problems by reducing cold groundwater flow from wet meadows to streams (Ponce and Lindquist, 1990).

Increases in channel width caused by high levels of sediment delivery and/or loss of bank stability also exacerbates seasonal water temperature extremes in winter and summer. In summer, vegetation of a given height is less effective in shading wider channels. Wider channels also have a greater heat load under a fixed energy budget at wider channel widths because the increase in the stream surface area increases the heat load to the stream.

Although not well-documented in the scientific literature, reduced groundwater contributions during the summer low flow period have probably contributed significantly to elevated water temperatures. Groundwater is a source of cold water and is usually at the average annual air temperature (or about 40-50°F). In many watersheds, the road network intercepts and drains significant amounts of groundwater. Channel incision caused by grazing has dewatered and desiccated many riparian wetlands in the Snake River Basin (Rhodes et al., 1993). Roads, grazing, logging, and mining have severely compacted and disrupted wetland soils and vegetation, reducing the ability of the wetlands to store and release cold water during the summer baseflow period. This has also reduced baseflows and rendered streams more susceptible to rapid warming.

The sensitivity of summer water temperatures to changes in groundwater inflow, shading, and channel width is conceptually illustrated in Figure 27. Figure 27 displays the results of water temperature modeling via the USFWS water temperature model (Theurer et al., 1984) for a 20 cfs stream single trunk stream under a climatic conditions representative of the Blue Mountains with various combinations of shading, channel width, and groundwater input. Stream conditions used for the four modeled scenarios are contained in the caption for Figure 27. The rate of increase in temperature from the origin, which starts with 52°F stream water temperature, was greatest for streams with high width-to-depth ratios (W/D), low groundwater inputs, and low levels of shade. It appears that the influence of the lower rate of lateral inflow for the base case causes its sharper rate of increase than in the "low shade" case. The "high W/D" case has a W/D that is 9 times greater and a shade level that is more than 2 times greater than the "low shade" case. Even so, these cases had nearly the same rate of heating. The "base" case had the same W/D as the "low shade" case but had a significantly lower rate of heating, due to 80% shading under the "base" case and 30% shade under the "low shade" case. Comparing the "base" case with the "high W/D" illustrates the effect of W/D. The "high W/D" case has a W/D that is approximately 9 times greater than the base case and only a slightly lower shade value. Similar large increases in stream heating can then be accomplished by increasing channel W/D or by removing shade. In the field, both effects can occur by removing riparian trees.

Average daily stream temperatures reach 68°F in the three alternatives between 15.6 to 20 miles downstream of the origin, whereas it takes 40.3 miles of flow to reach the same temperature under conditions provided in the "base" case. The "base" case scenario provides about 20-24.4 miles more usable rearing habitat than is provided by the other scenarios, using a mean daily temperature of 68°F as a temperature threshold (See Figure 27). This figure also provides an indication of the amount of suitable rearing habitat that can be lost due to water temperature elevation.

While increased water temperatures on a single stream within a watershed may not significantly reduce production on the watershed level, the combined effects of temperature elevation can result in the loss of significant amounts of downstream rearing areas (Everest et al., 1985; Beschta et al., 1987; Hostetler, 1991). It appears that all vegetation within approximately 60 to 130 feet on either side of streams must be retained in forested environments in order to maintain the shading provided by old-growth forests; this range in vegetation width will vary with stream aspect, topography, vegetation type, and channel width (Beschta et al., 1987).

Losses in stream shading are significant because the recovery of water temperature and stream shading after logging takes about 25 years in the western Cascades (Gregory and Ashkenas, 1990). In many parts of the Snake River basin, the recovery of water temperatures and shading may take even longer because re-growth is slower (e.g., conditions are frequently harsh at higher elevations). In some watersheds key riparian species such as willows have been depleted for so long by livestock grazing that natural dispersal of colonizing vegetation will entail prolonged recovery horizons. However, vegetative recovery in riparian areas can only occur once impacts to riparian vegetation have been arrested. Vegetational shading and water temperatures will never recover as long as roads, mines, or poor grazing systems remain active in riparian zones.

1.3.1.3 Evaluation: Although the relationships among land use, water temperature, and salmon production are clear, sound management aimed at protecting water temperatures in salmon habitat has not been implemented. Streams throughout much of the Snake River Basin have had streamside vegetation stripped by logging, grazing, agriculture, road construction and mining (Theurer et al., 1985; ODEQ, 1989; ODFW et al., 1990; Nez Perce Tribe and IDFG, 1990, Anderson et al., 1992 (See Figure 28)). Many streams in the Snake River Basin with stream temperatures that naturally exceed levels optimal for salmon have had water temperatures elevated still further (See Figures 25 and 29); this has greatly reduced the production potential and the amount of usable rearing habitat in these streams. Most grazed riparian areas throughout the western U.S. have significantly less stream surface shade and higher summer water temperatures than non-grazed stream corridors (Platts, 1991). Stream discharge during the low flow period has been reduced by irrigation withdrawals and it is likely that grazing has exacerbated low flow problems (Ponce and Lindquist, 1990). Pool volume has been lost (Sedell and Everest, 1990; McIntosh et al., 1994) and streams are wider rendering streams more susceptible to temperature extremes. Adversely high water temperatures are now endemic in watersheds with significant logging, grazing, mining, and agriculture. Many streams that once supplied productive chinook habitat are now significantly impaired or even barren due to high summer stream temperatures. In their study on the Tucannon River, Theurer et al. (1985) concluded that the current summer temperature regimes, alone, could account for the reductions in chinook populations in the watershed that have been documented over the past several decades.

Water temperatures in historic and much current natal habitat in the Tucannon, Grande Ronde tributaries, and lower Salmon river commonly exceed levels that are directly lethal to salmon. Given the current condition of water temperatures and riparian vegetation in much of the Snake River basin, restoration of riparian vegetation is vital.

Available information indicates that the elevation of summer water temperatures impair salmon production at scales ranging from the reach to the stream network and put fish at greater risk through a variety of effects that operate at scales ranging from the individual organism to the aquatic community level. Increases in maximum summer water temperatures above 59^{o}F progressively impair salmon production. However, many smaller streams naturally have much lower temperatures and these conditions are critical to maintaining downstream water temperatures. At the stream system level, elevated water temperatures reduce the area of usable habitat during the summer. Loss of lower river rearing habitat from water temperature increases may have resulted in reduction in life history diversity of many salmonid species over the period of development in many basins caused by the loss of segments of the population having life histories that exploited lower river habitats during spawning or rearing. In some cases, the most potentially productive and structurally complex habitats, occurring in larger reaches, are rendered unusable due to water temperature elevation. Decreases in winter water temperatures also put salmon at additional risk. The loss of riparian vegetation, channel widening, and reduced baseflows exacerbate seasonal water temperature extremes. Elevated summer water temperatures also reduce the diversity of coldwater fish assemblages.

We recommend that activities that have the potential to increase water temperatures should not be allowed on any stream. We recommend that watersheds should be managed to increase the downstream extent of summer water temperatures that are suitable to salmon. We recommend that where daily maximum water temperatures in excess of 60°F exist in salmon habitat, measures should be taken to reduce water temperatures and activities that can forestall the recovery of natural stream temperatures should be prohibited or discontinued. Efforts to restore natural water temperature regimes should focus on the restoration and protection of riparian vegetation and hydrologic regimes.

Loss of vegetative shading is the predominant cause of elevated water temperature increases in natal habitat. Activities that decrease shading or forestall the recovery of shading should not be allowed. In streams draining managed watersheds, an increasing trend in shading should occur. We recommend the establishment and protection of riparian reserves in lieu of a stream shading standard (See Section 3.1 Riparian Reserves).

1.3.2 Miscellaneous Pollutants

1.3.2.1 Effects on salmon: Salmon require water that is free of pollutants. Chemical pollution in natal habitat can affect all freshwater lifestages. In general, chemical pollution is a density-independent source of mortality. A wide variety of dissolved mineral and chemical pollutants wreak havoc on all lifestages of salmon (USEPA, 1986). Both chronic and episodic (spills) chemical pollution can result in complete mortality of all fish in the natal habitat. Full listing and discussion of these chemical constituents is beyond the scope of this review, but the reader is referred to USEPA (1986).

1.3.2.2 Activities affecting pollutant loads: A wide variety of activities are point and nonpoint sources of chemical pollutants. The most significant sources in natal habitat are mining, crop agriculture (USGAO, 1990; Nelson et al., 1991), and chemical spills from toxics transported along natal streams. Return flows from agriculture can cause considerable direct mortality in chinook salmon (Saiki et al., 1992).

Acid mine drainage is generally the most serious and pervasive pollution caused by mining. Acid mine drainage has toxic, acute and chronic effects on fish due to acidity and dissolved metals associated with heavy metals (Nelson et al., 1991). Mining has already considerably polluted the water of the U.S. and it is estimated that more than 10 million fish were killed in the U.S. from 1961 to 1975 due to mining-related pollution (Nelson et al., 1991).

Recent spills during the transportation of toxic chemicals in the North Fork of the John Day River in Oregon and in the Shasta River in California indicate that the spills can cause high levels mortality of salmonids. These incidents clearly indicate the potential hazards to water quality in natal habitat by transporting toxics along streams. The spill of toxic chemicals could cause the complete mortality of at least two age classes of salmon, if a spill occurred during the spawning period. Mining activities have greatly increased the likelihood of toxic chemical spills in many parts of the Snake River Basin due to increased transportation of chemicals associated with the mines.

The transport of toxic chemicals along streams is a significant threat to the continued presence of summer and spring chinook in many drainages. For instance, the Salmon National Forest (SNF) (1991) estimated that the annual probability of a cyanide spill will be about 0.05% once the Beartrack mine operation is underway; given that probability, a toxic cyanide spill can be anticipated about every 230 years (SNF, 1991). During the next 50 years, the odds are approximately even that a petroleum spill will occur due to the project; there is about a 22% probability that a cyanide spill will occur during the next 50 years (SNF, 1991). In aggregate, due to the implementation of this project and resultant transportation of chemicals, there is a combined probability of 64% that a spill of either petroleum or cyanide will occur at least once during the next 50 years in the tributaries of the Salmon River due solely to chemicals transported for single mining operation. The SNF (1991) concedes that a spill will likely be toxic to fish. There are a significant number of mining operations occurring in the Salmon and Clearwater River Basins. The existing probability of a chemical spill from transported toxics is not known in most watersheds. However, main rail and truck transportation arteries parallel natal streams and a significant risk does exist in most streams.

Salmon habitat has been rendered unusable by acid mine drainage in Panther Creek, a tributary of the Salmon River; the spawning runs in the drainage were decimated by the pollution (Nelson et al., 1991). Cyanide heap leaching also poses a considerable threat to chinook (Nelson et al., 1991).

1.3.2.3 Evaluation: Many water quality pollutants affect salmon survival negatively. Episodic or chronic pollution has the potential to nullify all other efforts to protect and restore salmon habitat and populations. We recommend that major efforts should be initiated and maintained to prevent water pollution. Water quality should be monitored closely in watersheds where there is potential for pollution of water quality in natal habitat. Although state and federal water quality standards may not adequately protect salmon from the adverse effects of chemical pollutants, these standards do represent minimum protection levels and should be monitored, met, and enforced. We recommend that state and federal water quality standards should be reviewed to determine their efficacy in protecting salmon from increased mortality; new or revised water quality standards should be adopted as needed to protect salmon survival.

However, monitoring can only detect problems *ex post facto*. As with other habitat conditions, we recommend that measures to prevent chemical pollution should be a high priority. Given the inherent risk of spills, we recommend that toxic materials should not be stored in watersheds that provide habitat for the listed salmon species. We also recommend that the transport of toxic chemicals along spawning and rearing habitat and their tributary streams be eliminated or restricted. We also recommend that all existing and planned mining operations be thoroughly evaluated for their potential for giving rise to acid mine drainage.

1.4 WATER QUANTITY AND TIMING

1.4.1 Effects on salmon: Stream discharge strongly influences almost every aspect of natal habitat. It is one of the major controls on the space available for spawning and rearing (Bjornn and Reiser, 1991). The frequency and magnitude of stream discharge exerts a strong control on substrate and channel morphology (Dunne and Leopold, 1978; Richards, 1982; Swanston, 1991; Chamberlin et al., 1991). Increased peakflows can cause redd scouring and/or channel widening and incisement which can cause downstream sedimentation and habitat degradation (Richards, 1982; Chamberlin et al., 1991; Furniss et al., 1991; MacDonald et al., 1991). In the Idaho batholith, sediment delivery increases with increasing peakflow (Megahan and Bohn, 1989). Winter and summer flows also exert a profound influence on temperature extremes; lower flows are more susceptible to seasonal temperature extremes in both winter and summer (Beschta et al., 1987). Adequate flow depths are also critical to the passage of returning adult salmon (Bjornn and Reiser, 1991). In the Umatilla River in Oregon, where water availability is a major problem, flows during the spawning and rearing period appear to exert a profound influence on the number of returning steelhead (Confederated Tribes of the Umatilla Indian Reservation, 1994 (See Figure 30)).

Shallow groundwater is an important source of streamflow in many alluviated systems. Groundwater is also important for temperature mediation. Groundwater inflows typically cool streams during the summer and warm them during winter.

Water flows are also critical to migrating salmon in the mainstem. The best available data indicate that smolt survival downstream increases with increasing water discharge (Cada et al., 1994). Higher water discharges also appear to improve the migration conditions for fall chinook by improving water temperatures (M. Karr, CRITFC Fish Passage Specialist, pers. comm., 1993).

1.4.2 Activities affecting water quantity and timing: Water withdrawal reduces all flows downstream of the point of withdrawal. Due to seasonal demands, water withdrawals in the Snake River Basin are typically greatest during seasonal low flow periods in the summer. Logging and roads in snow-dominated climates can increase peakflow (King and Tennyson, 1984; King, 1989; MacDonald and Ritland, 1989; Harr and Coffin, 1990; Chamberlin et al., 1991). However, logging also typically increases summer baseflows when significant portions of a watershed have been deforested (Bosch and Hewlett, 1982; Chamberlin et al., 1991). This increase in summer low flow is typically shortlived as vegetational regrowth occurs (Chamberlin et al., 1991). Logging and road construction also cause annual peakflows to occur earlier in the snowmelt period.

Grazing increases peakflows (Platts, 1991), while decreasing low flows. Low flows are decreased by grazing primarily through the de-watering of wet meadows by incised channels, increased overland flow caused by compacted soils, and reduced water storage in compacted soils. Field evaluations indicate that channel incisement caused by grazing has reduced baseflow contributions during the low flow period in many meadow systems (Beschta et al., 1991; Beschta et al., 1993). Improved grazing management is probably the most promising land management strategy available for increase summer baseflows (Ponce and Lindquist, 1990).

Mining and crop agriculture increase peakflows (Dunne and Leopold, 1978; Richards, 1982). These same activities also typically withdraw water during the summer which reduces low flows.

It is widely acknowledged that seasonally-saturated and perennially-saturated riparian areas, wetlands and springs are vital to the maintenance of low flows during winter and summer (Ponce and

Lindquist, 1990; Gregory and Ashkenas, 1990; Naiman et al., 1992). Mining, road construction, grazing, and tractor logging in wetlands disrupt wetland functions and reduce low flow contributions.

Roadcuts have been documented to intercept subsurface water in the Idaho batholith (Megahan, 1972). This interception may increase peakflows while decreasing subsurface baseflows to streams (Megahan, 1972). It appears that roadcuts will always intercept subsurface flows when they cross hillslopes that carry subsurface flows to streams (Kirkby, 1978).

Groundwater pumping in alluviated valleys or in areas where groundwater feeds streamflows can significantly alter streamflow, leading to decreased baseflow (Freeze and Cherry, 1979). In some cases, groundwater pumping can seasonally dewater stream segments, blocking salmon passage into spawning habitat. The loss of groundwater inflows can intensify seasonal temperature extremes as well as reduce streamflow.

Water withdrawals in Snake River Basin tributaries have undoubtedly cumulatively reduced downstream water supply and availability on the mainstem Columbia River.

1.4.3 Evaluation: Many stream reaches with historic natal habitat in the Snake River Basin have flows reduced significantly during the summer low flow period. Many riparian wetlands, such as wet meadows, have been damaged by grazing, mining, road construction, and logging in the Snake River Basin as consistently indicated by field reviews (Beschta et al., 1991; Beschta et al., 1993). This loss of wetland function has probably contributed to reducing summer low flows in natal habitat.

Although data are sparse, peakflows may occur a week or two earlier in the year in some managed watersheds year than in unmanaged watersheds. McIntosh (1992) found that the annual peakflows currently occur about 2 weeks earlier in the Grande Ronde than historically.

Some heavily logged drainages may have increased summer low flows; summer low flow has increased in the some parts of the Grande Ronde over the past 50 years (McIntosh, 1992). However, the increases in low flows do not appear to have improved salmonid survival because the water quality is so poor and stream habitats have been heavily degraded (B. McIntosh, USFS PNW Research Station Res. Asst., pers. comm., 1992; J. Sedell, USFS PNW Research Station Aquatic Ecologist, pers. comm., 1993) due to upstream logging, grazing, and road construction (Anderson et al., 1993; McIntosh et al., 1994).

Although we cannot currently recommend numeric standards for seasonal flows, it is apparent that flow decreases during the low flow period can have several negative effects on natal salmon habitat and may also constrain mainstem options. Additional reductions in water discharge are inconsistent with efforts to improve habitat conditions and salmon survival. Therefore, we recommend that no additional withdrawals of surface water or groundwater should occur in any watersheds with salmon habitat or tributary to waterways that provide salmon habitat, until it is documented that resultant flows will be adequate for salmon survival and the maintenance or restoration of favorable habitat conditions. Studies should be undertaken to determine the instream flow levels needed for salmon in each watershed. These studies should include assessment of the seasonal streamflows needed to maintain channel morphology, sediment routing, floodplain function, and adequate water temperatures, as well as the streamflows needed for salmon passage, rearing, and spawning; broad scale studies should also determine the cumulative effects of water withdrawals in tributaries on mainstem flow options. Where current instream flows are inadequate for salmon or for the restoration and maintenance of favorable habitat conditions, we recommend that efforts should be made to acquire water for instream salmon and habitat needs.

We also recommend that all wetlands be fully protected from adverse soil and vegetation impacts. Efforts should be made to restore baseflow regimes by restoring the function of degraded meadow systems. Implementation of our recommendations on constraining vegetation removal at the watershed scale based on sediment delivery, together with establishment of riparian reserves, protection of roadless areas, and suspension and alteration of riparian grazing, and reductions in road mileage should provide some protection against increased peakflows and decreased baseflows.

1.5 THE USE OF "RANGES OF NATURAL VARIABILITY" AS AN ALTERNATIVE APPROACH TO THE DEVELOPMENT OF HABITAT STANDARDS

We recommend that a single set of habitat standards be applied across the Snake River Basin for all habitats, based on biological habitat requirements of salmon. Data do not indicate that the biological requirements of salmon vary significantly among regions. The purposes of the standards are to gage the likely effect of activities on habitat conditions and salmon populations and to trigger management responses needed to restore and protect salmon habitat in altered landscapes consistent with efforts to recover listed salmon populations. This approach has the most promise for habitat protection and species recovery because it does <u>not</u> rest solely on statistical approaches to setting habitat standards.

Other researchers have labeled our approach as "one-size-fits-all" (Peterson et al., 1992). For brevity, we refer to our approach as a "biologically-based approach." Some (USFS et al., 1993; Wissmar, 1993) have charged that use a single set of numeric standards across a region fail to address issues of attainability of the habitat standards in specific watersheds and the natural variability of habitat attributes within complex stream ecosystems (USFS et al., 1993; Wissmar, 1993). Even those that have proposed a hybridized variant of the "one-size-fits-all" approach have voiced some similar reservations about it (Peterson et al., 1992). However, we believe that upon examination, other alternative approaches have the same pitfalls, considerably greater operational complexities, some logistical impossibilities, and much less promise as a tool to facilitate habitat assessment and management that is aimed at protection and restoration.

A prominent alternative approach to the biologically-based approach is to set standards for habitat and landscape attributes based on a sampled range of natural variability in various regions or ecological watershed complexes (USFS et al., 1993; Wissmar, 1993). For brevity, this approach will be referred to generically as the "Range of Natural Variability Approach" (RNVA) despite the minor variations peculiar to individual formulations. Although proponents of the RNVA (USFS et al., 1993; Wissmar, 1993) provide little practical detail on the methods necessary for implementation of the RNVA, it appears that the RNVA requires sampling habitat parameters (e.g., LWD frequency) or landscape parameters (e.g., the fraction of watershed vegetation in various seral states) in relatively unimpacted watersheds within somewhat homogeneous biogeoclimatic regimes. Habitat and/or watershed standards are then set as ranges on regional or watershed bases. Both USFS et al. (1993) and Wissmar (1993) allude to the possibility of using theoretical models, as well as data, to generate quantifiable standards and/or statistically determined ranges; however, no other specifics about modeling approaches are given. Peterson et al. (1992) present a hybrid of the two approaches: a single set of target conditions based on existing data from undisturbed watersheds in western Washington.

The primary assumption inherent in the RNVA is that "healthy" watersheds with "healthy" fish populations have habitat conditions that vary naturally in time and space and that habitat conditions vary naturally among watersheds, and vary among reaches within watersheds (USFS et al., 1993; Wissmar, 1993). Available data adequately indicate that landscape, riparian, and aquatic habitat conditions naturally vary in time and space within and among watersheds. However, data are extremely limited on the frequency and duration of various states of habitat conditions in natural systems. Data do not generally exist to clearly delineate what the full effects of this variation are on fish populations. However, available information indicates that cycles of variation do influence salmon survival and population levels. For instance, it is probable that fires, floods, and other climatic cycles can affect habitat conditions and salmon populations.

1.5.1 Evaluation: Given the current data limitations, critical uncertainties, and untested and undeveloped methodologies, the RNVA provides a cumbersome process for developing standards that have just as much uncertainty regarding attainability and variability as the "one-size-fits-all" approach, but with much more analytic effort and less promise for protection.

The RNVA has limited promise for the protection of fragile resources such as weak populations of salmon that are geographically isolated. Due to its reliance on statistical detection of significant change in highly variable phenomena, the RNVA inherently allows degraded conditions to persist until either the effect is pronounced and persistent or a large sampling effort is complete. Given inherent variability in environmental phenomena, statistically significant degradation of many resources may not be detected until damage is severe and/or irreversible (Ludwig et al., 1993); in such cases, sole reliance on statistical methods of detection prior to changing management is bound to fail to protect fragile resources (Ludwig et al., 1993). The listed salmon and their habitats are just such cases. These inherent problems are exacerbated because the RNVA only triggers changes in land management after ranges of habitat conditions have been exceeded; they also ignore the biological consequences to salmon populations of allowing degraded habitat conditions to persist just because they are within sampled ranges.

The RNVA does not address the habitat conditions needed to allow salmon recovery. Instead, the RNVA deems habitat conditions acceptable as long as they fall within statistically determined ranges (Wissmar, 1993), even though the conditions may be neither natural (in absolute value or in terms of frequency of occurrence in time or space) nor conducive to salmon survival. Under this scenario, it is possible to have all streams in a region with habitat conditions that are not conducive to salmon survival, without triggering changes in land management, because the conditions are within sampled ranges assumed to represent natural conditions. The RNVA sets ranges for habitat attributes that potentially include conditions caused by natural catastrophes in natural landscapes (flood, fires, episodic landslides, etc.) that negatively affect salmon survival and production for a period of time. This greatly weakens the potential of the approach as a protection and evaluation tool, because degraded conditions caused by land management may be within the bounds of conditions caused by natural catastrophes that occur with low frequency.

The ramifications of statistical approaches to protecting fragile resources through detection of change are worth noting, although a full review of these statistics is beyond the scope of this report. The magnitude of a minimum detectable effect (MDE) at a given level of statistical significance increases with increasing variability or decreasing sample size (Peterman, 1990). For instance, Lichatowich and Cramer (1979) found that studies of survival and abundance of salmon populations may require 20 to 30 years of sampling to produce an 80% chance of detecting a 50% change.

Statistical approaches inherently have the risk of producing Type II errors. A Type II error is committed when one concludes that there has been no change in a parameter when, in fact, there has been a change (Benjamin and Cornell, 1970). Type I errors are committed when one concludes that there has been a change in a parameter when, in fact, there has been no change. The RNVA focuses solely on Type I issues and completely ignores Type II errors, even though Type II errors are often more costly (costs incurred to date, plus the costs of remedial action in the future) than Type I errors, especially with fragile resources where effects are not quickly reversible (Peterman, 1990). The probability of making Type II errors becomes greater with increased variability and decreased sample size. The propensity of an RNVA to make Type II errors is exacerbated because data are limited on frequency, duration, and extent of habitat conditions from stream systems that could be potentially considered representative. Notably, proponents of the RNVA (USFS et al., 1993; Wissmar, 1993) completely fail to incorporate the costs or probability of making Type II errors in assessing habitat conditions that affect survival of salmonids that are at risk of extirpation. In a review of the ramifications of statistical power and Type II errors, Peterman (1990) concluded:

> "Scientists should explicitly report the biological ramifications of Type II errors...Scientists *must not* assert, either explicitly or implicitly, that 'no effect' has occurred when a data analysis fails to reject some H_O [the hypothesis that an effect has not occurred, e.g., that land management has not reduced survival]. They must also not draw conclusions or make recommendations based on failing to reject H_O unless there is high statistical power...This practice *must* change if scientists are to improve the quality of interpretations of their results." (Emphasis is his; bracketed material was added).

Unfortunately, the RNVA disregards every aspect of the advice of Peterman. The RNVA does not require high statistical power, nor does it include explicit consideration of the biological ramifications of Type II errors.

The biological ramifications of maintaining degraded habitat conditions because they are within the range of natural variability are likely to be significant. As an example, consider a hypothetical case where the MDE of management on surface fine sediment is an increase from a mean of 20% surface fine sediment to a mean of 30% surface fine sediment due to sample size, variability, and given level of statistical significance. Suppose that monitoring indicates an increase from 19% surface fine sediment to 28% in a hypothetical stream and that the condition persists for ten years with no change in management. Based on regression analysis of the data of Scully and Petrosky (1991), egg-to-parr survival would have dropped from about 37% to 25%; thus, survival at the egg-to-parr stage would have been cut by about a third of that initially existing. The effect would have persisted for a decade without any effort made to reverse the effect and it could be expected that the condition would further persist even if action were taken due to in-channel lag times and the low reversibility of on-site impacts. It is probable under an RNVA that changes in land management would not occur in the hypothetical example until surface fine sediment increased further and salmon survival was further decreased; management options may have also been foreclosed by continuing expansion of logging and road construction in a watershed with a stream that is within the "range of natural variability." This simple example clearly indicates that application of an RNVA leads to results that are counter to protecting and restoring salmon habitat, survival, and populations.

The RNVA is premised on the false notion that conditions within specified ranges indicate that habitats have not been degraded by management (Wissmar, 1993). The RNVA ignores causative linkages between landscape alteration and habitat conditions. Application of this approach in other venues would lead one to conclude that all deaths in a given human population were natural as long as an individual's age at the time of death was within some specified fraction of a standard deviation of the mean human lifespan, regardless of actual causes. Available information clearly indicates that in managed watersheds, watershed functions will be altered and that this will lead to habitat alteration, albeit, in some fashion that may not be completely predictable. Current knowledge and understanding of watershed and ecological processes indicate that altered landscapes always lead to altered habitat conditions; that the alteration is within the range found in other systems does not and cannot mean it is in natural condition, only that it is deemed acceptable within the framework of an RNVA.

The RNVA proposed by Wissmar (1993) fails to recognize that the mimicry of natural variability in habitat conditions in managed systems requires that these conditions not only be within some statistically determined range found via sampling, but also at the natural <u>frequency and duration</u> within individual watersheds and among populations of watersheds. In a given watershed, maintenance of LWD at levels well below the sample mean but within sampled ranges for two centuries in a given watershed does not mimic the variability of natural systems where it would be expected that LWD levels would oscillate about the mean over time. Further, maintaining LWD levels in many managed watersheds below a sample mean but within sampled ranges over time probably also fails to mimic the historic spatial variability among watersheds over time. Thus, as presently conceived, the variant of the RNVA proposed by Wissmar (1993) fails to recognize considerations of spatial and temporal frequency and duration must be used to assess whether managed systems appear to mimic the variability found in natural systems.

The variant of the RNVA proposed by USFS et al. (1993) conceptually addresses the frequency and duration issues by calling for restoration of the frequency and duration of natural processes operating in natural watersheds. However, USFS et al. (1993) fails to note that this can only happen once watersheds have returned to some natural condition. USFS et al. (1993) tacitly asserts that restoration of natural functions, natural conditions, and natural variability can be restored in altered landscapes as long as some portions of the landscapes are protected. The assumption that natural function and variability can be restored by restoring and protecting only part of an ecosystem runs counter to most current knowledge of the integrated nature of hydrologic and biologic systems. Under the RNVA variant proposed in USFS et al. (1993), no control criteria or methodology are given by which to assess when and if natural processes are in operation. It is only assumed that natural conditions will occur at natural frequencies as long as key portions of the landscapes are protected, even though this may not occur.

Although the purported ability of the RNVA to address issues of attainability and variability makes it appear attractive at first glance, there are a number of formidable barriers to applying a RNVA. Data on the frequency, duration, and extent of habitat conditions is currently lacking.

It may not be possible to collect applicable data for many systems because very few large, low gradient streams have habitat conditions that reflect solely natural conditions. Streams with natural conditions may not be representative of degraded streams. Notably, larger order, low-gradient streams are typically the most potentially productive and structurally diverse habitats (Frissell, 1992). Unfortunately, the same habitats are the most sensitive to cumulative degradation, especially by sedimentation, due to their reach characteristics and position within watersheds (Frissell, 1992; Rosgen, 1993). Few larger order, low elevation streams are in natural condition.

The array of potential choices of less-disturbed watersheds in the Blue Mountain Province of the Snake River Basin underscores data availability problems associated with the RNVA. Only the North Fork of the Umatilla River, the North Fork of the John Day (NFJDR), the Wenaha, the Minam, the upper Imnaha, and the upper Tucannon rivers within their wilderness portions could hold promise for yielding data that could potentially be construed to be indicative of natural conditions. However, all of the six watersheds have had varying levels of impacts. The NFJDR has tributaries outside of the roadless/wilderness areas that have been logged, roaded and mined. The Minam was splash-dammed. The Imnaha headwaters were historically subjected to heavy sheep grazing. The fringes of the Wenaha watershed contain tilled agriculture. All of the areas historically have been subjected to varying levels of livestock grazing. The level of these impacts and resultant deviation from natural function is unknown, as is the potential level of recovery from the impacts. Thus, it is extremely doubtful that data from these less perturbed systems could provide any reasonable indication of natural ranges or habitat attributes or the level of attainability in damaged systems, even if other existing complicating factors did not exist.

It is doubtful that "less-perturbed" watersheds can be used as a surrogate for undisturbed basins in collecting data to determine natural ranges of variation for factors such as discharge or sedimentation. Attributes that are affected by sedimentation and discharge regimes (channel morphology, water temperature, pool volumes, substrate conditions, etc.) in "slightly" perturbed systems probably depart from natural conditions by some unknown magnitude which may vary in a non-systematic or unknown fashion among perturbed watersheds; such data cannot be construed as a measure of natural conditions or ranges. The uncertainty associated with the magnitude of departure from natural conditions and temporal frequency of natural conditions is compounded by uncertainty in lag times for both reaction to perturbations and recovery from perturbation. Data from watersheds affected by sedimentation could be representative of a reaction period associated with channel adjustment to changes in discharge/sediment delivery, a relaxation period of equilibrium with existing perturbation levels, or a recovery period after cessation of the perturbation. However, methods do not currently exist for delineating these potential states. It appears that changes in channel morphology, sediment transport, bed composition, caused by sedimentation may persist in channels for more than 100 years (Platts et al., 1989; James, 1989) depending on the magnitude and persistence of increased sediment delivery, channel discharge regime, and other factors. Therefore, such data from less-perturbed systems are indeterminate and cannot be used to estimate natural conditions or ranges in habitat or watershed attributes. Thus, although the RNVA has theoretical allure, existing conditions and uncertainties render it unusable. Given existing watershed conditions, known linkages causing unknown departures, data availability, probable variability, and critical uncertainties, it cannot be assumed that the spatial and temporal variability and range of habitat conditions can be currently estimated. The RNVA will falter for lack of data to adequately characterize the information that is at the core of the approach.

Even lacking a meaningful classification and stratification regime, it is clear that the few undisturbed, or less disturbed, watersheds remaining are **not** representative of conditions once existing in the vast number of degraded watersheds. Most remaining roadless areas within the Snake River Basin are in areas of higher elevation, higher topographic relief, higher precipitation, colder climates, more alpine vegetation, higher gradient streams, less stable soils, hydrology more dominated by snowmelt, and with a higher propensity for erosion than the many degraded systems that currently host the listed species. Due to these conditions, it is likely that these high elevation and high relief roadless watersheds can be expected to have fewer pools, more fine sediment, and less and smaller wood in their natural state than found in the lower elevation systems with more stable soils in their natural state. As mentioned, the problem of the lack of representative watersheds is compounded because the most productive reach types for salmon habitat (large order, low gradient, unconstrained streams) do not exist in their natural state, due to their locations within watersheds and their sensitivity to cumulative effects.

Consideration of the six least-perturbed streams in the Blue Mountain Province of the Snake River Basin mentioned above points out some of the likely obstacles to collecting the data needed to implement a RNVA. Data from six watersheds are unlikely to be statistically reliable. However, data from the six basins cannot be lumped, because they are considerably different from one another. All six basins have distinctly different geology and geomorphology. The North Fork of the John Day watershed is highly erosive and prone to mass failure. The North Fork of the Umatilla River is in a steeply incised basaltic canyon. The upper Imnaha and Minam have significant portions of their headwaters above treeline in glacially modified terrain. Most of the reaches within the six systems occupy canyons and are without large floodplains; the majority of reaches are constrained and of relatively high gradient. Even absent a classification scheme, it is extremely doubtful that data from these unperturbed or less perturbed systems could provide any reasonable indication of natural ranges or habitat attributes or the level of attainability in damaged systems. Again, while arguments for the RNVA are superficially compelling, pragmatic considerations and existing conditions make it unusable.

The RNVA inherently depends on valid watershed classification and the lack of validated watershed classification systems poses a serious obstacle to the RNVA. The RNVA requires either monitoring of habitat conditions over time or monitoring conditions in many watersheds as a "space for time" substitution for trend sampling in specific watersheds. The sampled ranges for conditions are then extrapolated to other watersheds, that are assumed to exhibit similar ranges of variation. Substitution of space for time in monitoring and extrapolation relies on valid watershed classification.

It is not currently possible to *a priori* determine that sampled watersheds have the same frequency distribution of the aquatic and terrestrial functions over time and space. This poses serious difficulties because, at a minimum, the RNVA requires that sampled watersheds have similar temporal and spatial distribution of habitat conditions expected in damaged watersheds under natural conditions. Empirically, it has been generally shown that adjacent watersheds with similar climates can have extremely dissimilar distributions of flow frequencies (Reich, 1977 as cited in Wood and Hebson, 1986). Systems with dissimilar flow frequencies also probably have much different distributions in the magnitude and frequency of sediment transport, storage, and sedimentation and, therefore, different ranges and frequencies of substrate and channel morphology conditions. Although methods have been proposed for ascertaining the similarity of probability distribution functions of discharge among watersheds (Wood and Hebson, 1986), these methods remain hypothetical and untested. Methods of classifying watersheds based on the expected frequency distribution of habitat conditions do not appear to be currently available; this hampers application of the RNVA.

There are also operational barriers to using an RNVA, even if scientific obstacles can be surmounted. The RNVA requires considerable monitoring of habitat conditions over time and space in order to estimate the natural frequency distribution of habitat conditions. Intensive monitoring is also required to estimate the spatial and temporal distribution of habitat conditions in managed watersheds. It appears unlikely that any agency will undertake the level of monitoring necessary to reasonably assess that perturbed systems have habitat conditions that are at the duration, frequency, and extent found in unperturbed systems.

Even if all other problems associated with the RNVA could be surmounted, the RNVA fails to incorporate considerations of what conditions are required biologically by the listed salmon. The RNVA accepts all conditions within some fixed range whether or not they are conducive to salmon survival and regardless of the level of management causation. Although this approach may be feasible for protecting robust salmon populations that have survival rates adequate to maintain the population, it will be unsuccessful for protecting and restoring the listed salmon stocks because they have extremely low survival and are in severe decline. For this reason, alone, it should not be used to develop habitat and watershed standards for the protection and restoration of the listed salmon species.

In application, the RNVA probably will result in maintaining watersheds with the greatest habitat potential in a somewhat degraded condition. This is because habitat conditions will be deemed acceptable and damaging land management allowed to continue as long as habitat conditions fall within measured ranges based on conditions in watersheds that have somewhat poor habitat conditions due to either their inherent potential or temporal status. Thus, some of the potentially best habitats are never allowed to attain full potential. Therefore, the RNVA never fully addresses attainability because it may not allow systems (especially the historically most productive, lower gradient systems) to fully recover as a means of assessing attainable states. Rather, it makes assumptions of attainability based on habitat condition baselines derived from the existing conditions in stream types that may not accurately reflect conditions attained historically; this is especially true for some of the more productive, unconstrained, higher order streams.

The biologically-based approach has several advantages over the RNVA. Admittedly, the biologically-based approach also has some of the same disadvantages, but they are arrived at with considerably less informational and operational cost. The most important advantage of the biologically-based approach over the RNVA is that it does address key biological habitat requirements of salmon.

Although proponents of the RNVA have asserted that rational standards cannot be set without considering the attainability of standards and the variability of habitat conditions within various ecologic systems (USFS et al., 1993), there is little merit to those assertions. The biologically-based approach does not prevent natural variation in stream conditions from occurring; it only mandates that anthropogenic contributions to variation be curtailed once stream conditions are at levels that can be anticipated to impair salmon survival and production.

The RNVA may not address attainability well because it may not require that land management change until habitat conditions are well outside of natural ranges. Rather than becoming lost in issues concerning detection of management induced levels of habitat damage, the biologically-based approach focuses on attaining key habitat conditions and detecting deviations from those conditions **regardless of the cause**. The biologically-based approach acknowledges that natural and anthropogenic perturbations have combined effects on habitat conditions and salmon survival. Where conditions that cause reduced salmon survival exist, activities that aggravate or prolong these conditions should be curtailed or eliminated. The biologically-based approach does not directly consider issues of attainability of the standards in all watersheds. It is likely that in some reaches of some watersheds deviations from the standards exist naturally; it is likely that some of the standards cannot be attained in still more reaches of some systems. Deviations from the standards should serve as a signal that the systems are naturally sensitive to perturbation; exacerbation of naturally marginal conditions may render such habitats inhospitable to salmon. For instance, in streams where desirable water temperatures are naturally exceeded, the proper response is to ensure that no risk of water temperature increase is allowed, rather than citing their existence as justification for allowing high water temperatures in degraded streams to be maintained by not altering land management. Naturally high water temperatures indicate that such systems are physically and biologically sensitive to alterations that can increase water temperature in marginal cold- or cool-water fish zones. Likewise, the natural existence of high levels of surface fines and low pool volumes indicates that the watershed is naturally prone to erosion and/or the stream system cannot fully transport fine sediment and is susceptible to any increase in sediment delivery. In such systems, increased sediment delivery should be avoided if salmon survival is to be protected.

It may not be possible to develop a uniform testable approach to ascertaining the attainability of habitat conditions. Fully controlled experiments with full replicates may not even be possible to design given fluctuations in independent variables such as climate, the existence of strong feedback loops among such factors as channel geometry, stream power, sediment transport of various particle sizes, and channel bed composition, and channel attributes that may exhibit hysteretic (non-unique functional) responses to perturbations dependent on current conditions and history. Indeed, some hypothetical notions about watershed function may only be "testable" through Bayesian analysis (McAllister and Peterman, 1992). Even then, results from Bayesian analysis are based on presumed effects and the lumping of data that should not be lumped into a single distribution (Beven, 1989); many have questioned the utility of Bayesian exercises (McAllister and Peterman, 1992).

At the heart of the biologically-based approach is the recognition that a reasonable amount of data exists that can be used to define the condition of habitat attributes conducive to salmon survival and production. These conditions can be described in a quantitative fashion that allows measurement and determination of resource status. Habitat standards are based on available information about the biological requirements of salmon, not on estimates of attainability or variability that are intrinsically dubious. Questions of attainability and variability are not ignored, but they are not incorporated into the development of the standard because tools and data do not currently exist to adequately address those questions.

Notably, a similar approach to the biologically-based approach was taken by the Technical Advisory Committee for the Triennial Review of the Water Temperature Standard for the State of Oregon (TAC). The TAC repeatedly considered various methods of using models and/or data to develop stratified water temperature standards for various permutations of regions, elevational bands, watersheds, reaches, and stream sizes. The TAC concluded that it was not possible to develop such standards for the following reasons: 1) available data generally did not represent natural conditions and departed from natural conditions by some unknown magnitude, probably in some unsystematic way; 2) available data could not taken as representative of any particular stratum because watershed processes controlling water temperatures were highly variable among watersheds; 3) inadequate data existed to evaluate variability in and among potential strata; 4) models were of questionable accuracy in larger watersheds or had data limitations; 5) application of such an approach would probably lead to inadequate protection of the sites that naturally had the lowest water temperatures. However, the TAC concluded that water temperatures could be identified above which there was an increasing level of impairment of fish production and survival. Notably, the TAC gave more consideration and discussion to stratified standards than any other single matter.

Other advantages of the biologically-based approach are that is does not require the development and testing of watershed classification schemes, extensive monitoring to assess degree of current approximation to natural variation or trends in variation, and subsequent analysis and statistical manipulation of monitoring results prior to implementation. Application of the biologically-based approach moots the operational and technical difficulties and potential errors associated with trying to determine if habitat conditions fall within expected "natural" ranges. The biologically-based approach accepts management-induced habitat conditions provided they are consistent with rebuilding the salmon populations. Where standards are not met, for whatever reason, the biologically-based approach requires that management activities that contribute to the maintenance or exacerbation of these conditions be eliminated. Although the biologically-based approach does have operational complexities, it is far less burdensome than full implementation of an RNVA.

The biologically-based approach and the RNVA both have the potential to prevent the full recovery of damaged watersheds and habitats, because once all habitat standards have been met, potentially damaging activities can occur. Habitat standards developed under either approach may not reflect the best habitat conditions attainable in some watersheds. However, similar to USFS et al. (1993), the biologically-based approach includes standards for land use to limit the potential for aquatic habitat damage.

We recommend the biologically-based approach because it is both a more tractable and a more effective vehicle for protecting and restoring salmon habitats and populations. We do not recommend that the RNVA be used to develop habitat standards for screening because it is not currently operational. Ultimately, the RNVA may be useful for habitat management issues where attainability and the detection of land management-induced changes in habitat condition are deemed more important than the protection of endemic salmon populations. For listed salmon species in the Snake River Basin, protection and restoration of salmon habitats and their populations far outweigh attainability and detection considerations.

The RNVA may ultimately have some promise only if existing obstacles to its implementation can be surmounted. We recommend that if an RNVA is pursued, it should incorporate all the recommendations of Peterman (1990) regarding statistical approaches to fishery management. Even then, data availability for some systems may be a roadblock to implementation. The RNVA will probably never be a desirable tool for managing highly valued resources that are at risk of irreversible loss, especially when those resources exhibit high variability.

1.6 NOTES ON STATISTICAL SIGNIFICANCE

It is prudent to require a high level of statistical significance in assuring that degraded habitat conditions have improved prior to initiating or continuing activities that have potential to cause degradation or forestall recovery. It is also prudent to set a lower level of statistical significance in detecting deteriorating conditions and making attendant management adjustments. This is because habitat degradation and its causes are only slowly reversible. The biological consequences of habitat damage may not be reversible due to the fragile status of salmon populations. Consequently, a risk adverse approach to statistical detection of different types of change in habitat conditions is advisable.

We recommend a fairly high level of statistical significance for detection of improving trends ($p<0.05$). At this level of statistical significance, there is a fairly high probability that habitat conditions have actually improved prior to initiating activities that can potentially forestall habitat recovery. We also recommend a fairly low level of statistical significance for the detection of deteriorating trends in habitat variables set as standards ($p<0.40$). This relatively low level of statistical significance decreases the magnitude of the MDE at a given sample size, level of variability and statistical power (Peterman, 1990) and is aimed at eliciting a prompt management response to deteriorating conditions before departures are considerable and, potentially, irreversible over the course of several salmon generations. We have also set these differing levels of significance based on considerations of the ecological costs associated with the different types of errors involved in accepting various false hypotheses. The most rational approach to setting levels of statistical significance and power is to base them on the expected costs associated with management decisions based on the errors of accepting different false hypotheses as true (Peterman, 1990). Unfortunately, it is not possible to completely capture economic and ecologic costs of associated with accepting false hypotheses regarding habitat improvement, stasis, or deterioration. However, it can be anticipated that the costs of allowing poor habitat conditions to be maintained or further deteriorate far outweigh the costs of continuing to take a risk adverse approach to habitat protection and improvement even after it may have improved (Peterman, 1990; MacDonald et al., 1991). The maintenance of poor habitat conditions is likely to contribute to the relatively rapid extirpation of isolated spawning populations of salmon and the on- and off-site effects of land disturbing activities are slowly reversible. These conditions warrant setting much different levels of statistical significance for detecting the two types of change: improvement and deterioration.

However, as mentioned, the magnitude of the MDE is also a function of variability, sample size, and statistical power, in addition to the specified level of statistical significance (Peterman, 1990; MacDonald et al., 1991). These factors affect each other and there are statistical compromises involved in trying to adjust each of these factors (MacDonald et al., 1991). Rather than specify all factors, we recommend that the factors be adjusted through sample design or the selection of power so that MDE for accepting that deterioration has occurred at $p<0.40$ is no greater than a 10% deterioration in the initial value of the variable. However, some variables, such as water temperature, can have much smaller MDEs at the recommended level of statistical significance; in the case of testing for deterioration, monitoring efforts should be designed to make the MDEs as small as possible.

We have not specified statistical approaches. A number of approaches may be considered including, but not limited to non-parametric tests, regression analysis, and analysis of variance.

2.0 THE ROLE OF PASSIVE AND ACTIVE RESTORATION IN IMPROVING HABITAT CONDITIONS

We have recommended that passive restoration be implemented in all systems that do not meet habitat standards. While passive restoration should occur in all watersheds, active restoration is also important in damaged watersheds. The role of passive and active restoration in habitat improvement has been well-stated by Kauffman et al. (1993):

The first logical step in any restoration effort is the removal of those anthropogenic perturbations that are the principal causes of decline in ecosystem function and salmonid populations. A number of examples exist where removal of the primary human disturbances have resulted in dramatic improvements in salmonid habitats. We define this as "passive restoration." Often this is the least expensive and only activity necessary to successfully achieve salmonid habitat restoration.

For example, in rangelands, the cessation of livestock grazing often results in a rapid recovery of riparian vegetation and channel diversity. Frequently this is the only barrier to habitat restoration. Human perturbations are variable depending upon the ecosystem. Other examples of "passive restoration" include the cessation of excessive irrigation withdrawals, no-harvest buffers of an ecologically sufficient size, cessation of farming within riparian zones, and the cessation of chemical pollution of a riverine system.

There are also scenarios where the removal of anthropogenic perturbations may not result in the desired condition. In many degraded stream reaches, the removal of the primary disturbances may achieve some success, but a continued presence of ecosystem limitations may prevent a complete recovery. This scenario is represented by the box labeled "new ecosystem equilibrium" in Figure [31]. It is at this point that an "active restoration" program will need to be implemented. Of paramount importance is that active restoration programs should <u>not</u> be implemented until the removal of anthropogenic perturbations (or passive restoration) has proven inadequate for recovery. The most common cause of project failure in stream restoration is the implementation of active restoration activities before primary anthropogenic activities have been stopped.

Active restoration is defined as those activities that encompass mechanical, chemical, or biological manipulations of the ecosystem in order to achieve the desired future condition. This includes the reintroduction of native species, structural habitat additions, the removal of existing impediments to recovery (roads, dams, migration barriers, or rip-rap), vegetation manipulations (such as juniper removal), the use of prescribed fire, and chemical manipulations (such as the use of fertilizers and herbicides). Hence, active restoration may involve stream, riparian or upland watershed manipulations.

Among the greatest dangers in active restoration programs is the misinterpretation of ecosystem needs. This includes manipulations designed to restore habitats or ecosystems that instead result in a further degradation of the ecosystem. Failures in instream fish habitat restoration projects commonly occur when resource managers fail to recognize the linkages between the aquatic biota, riparian zones, uplands, and hydrological and climatic properties of the ecosystem. These misinterpretations of ecosystem needs most commonly occur when restoration activities focus on physical changes of instream habitat (structural approaches) rather than the promotion of biological or ecological functions. For active restoration programs to be successful, they: (l) must be sustainable; (2) they must facilitate the functioning of natural ecosystem processes; and (3) they must reconnect the linkages between the aquatic, riparian, and upland environments. Physical additions and structural approaches to stream restoration rarely achieve these goals.

> Misinterpretation of ecosystem needs can result in restoration activities that permanently sever linkages between the aquatic and riparian systems and hinder other ecosystem functions. For example, the placement of rip-rap or hard structures frequently results in the conversion of potentially productive unconstrained reaches into greatly simplified constrained reaches. Because of their influences on local channel hydrodynamics, channel morphology and streambank surface cover, these activities can dramatically limit vegetation recovery in both the treated and downstream areas. Therefore, this activity results in a reduced vegetation recovery that will have long-term detrimental influences on riparian/aquatic interactions such as reduced shade, nutrient inputs and coarse woody debris inputs. Ecosystem productivity will then be limited. Other common misinterpretations include the addition of unnatural features, such as boulders or wood structures in meadow/stream systems, or the introduction of exotic plants and fish (e.g., non-native plants, small mouth bass, brook trout, etc.). If the restoration of native fish populations remains a high priority in the Columbia Basin, future restoration and enhancement projects must consider ecosystem needs and functions in the design of these projects. It is no longer appropriate to simply identify and implement structural targets. Instead, anthropogenic impacts causing degradation need to be removed followed by the restoration of biological, chemical, physical and hydrological functions.

We strongly concur with their assessment. While we recommend that activities that contribute to degradation should not be initiated or continued, some watersheds are so damaged that active restoration will be necessary for habitat improvement. Active restoration should focus on addressing the causes rather than symptoms of degradation. In watersheds with high sediment delivery and/or substrate conditions that exceed standards, active restoration should focus on reducing sediment loads from anthropogenic sources such as roads and mines, especially within riparian reserves; sediment budget approaches to reduction of sediment delivery are highly recommended. In systems with low levels of pools, LWD, or high water temperatures, active restoration within riparian zones should focus on re-establishing vegetation within the riparian reserves. Although some in-channel approaches to habitat improvement may have some promise for improving habitat conditions in some streams, they should not be considered a surrogate for passive and active restoration addressing the causes of degradation and aimed at restoring the ecosystem functions needed to maintain desirable habitat conditions.

3.0 PERFORMANCE STANDARDS FOR LAND MANAGEMENT

Sole reliance on the evaluation of habitat conditions in relation to standards is inadequate to protect and restore the habitat conditions necessary to recover listed salmon species because the cumulative effects of watershed disturbances are lagged and are often slowly reversible. Under such an approach, the long process of recovery of habitat conditions is initiated only after damage has occurred and has been documented (Peterson et al., 1992). The avoidance and prevention of habitat damage at the watershed scale is likely to be both more effective and less expensive than attempting to remedy habitat damage once it has occurred. The ineffectiveness of relying solely on the detection of habitat degradation to trigger changes in land management is compounded by the biological context. Weak salmon populations may never recover from further habitat damage. Even if salmon populations can recover from habitat damage, it is sure that recovery of populations will lag behind the recovery of degraded habitat. The recovery of degraded habitat is slow and is unlikely to begin unless the causes of degradation are adequately addressed. Therefore, it is critical to develop land management standards that have some promise of preventing habitat damage and can lead to habitat recovery, as many other assessments have concluded (Anderson et al., 1992; Peterson et al., 1992; USFS et al., 1993; Henjum et al., 1994). These land use standards should be met even when in-channel standards are met until it has been documented that there has been recovery in the majority of habitats for other streams in the basin.

We have evaluated the utility of several land use standards. We recommend several as part of the screening process to evaluate the consistency of on-going and proposed activities with efforts to improve habitat and salmon survival. Activities that do not comply with the recommended land use standards should be considered inconsistent with habitat protection and improvement efforts. Some approaches to risk avoidance that are currently in use in the Snake River Basin were evaluated but are not recommended as part of the screening process.

We caution that there is considerable uncertainty regarding the effectiveness of our recommended standards. Although we have tried to incorporate some safety factors into some of the recommendations, there are very limited data indicating that habitat conditions can recover over time with on-going or existing landscape disturbance in a given watershed even with concurrent watershed restoration. Again, this underscores that habitat conditions and trend data are the bottom line. If monitoring does not indicate improvement in the habitat conditions set as standards, additional revision in land management should be implemented immediately, even when land management complies with the land use/watershed standards recommended here.

3.1 RIPARIAN RESERVES

3.1.1. Effects on salmon: A vast amount of data, studies, and information plainly show that riparian zones provide a variety of functions essential to the maintenance of habitat conditions conducive to salmon survival. The importance of riparian areas to water quality and habitat structure is greatly disproportionate to the percentage of the land base they occupy.

Riparian vegetation limits erosion and sediment delivery to salmon habitat through a variety of mechanisms. The loss of riparian vegetation can greatly increase sediment delivery by increasing rates of mass, surface, and fluvial channel erosion and reducing sediment detention within terrestrial and aquatic zones. Riparian vegetation provides shade and thermal cover, moderating temperature extremes in summer and winter. Riparian vegetation also provides a source of LWD and bank stability from deep rooted vegetation that is vital for creating and maintaining channel complexity, sediment storage sites, large pools, and cover. Riparian and floodplain zones provide a variety of important functions, such as groundwater storage, baseflow maintenance, and floodwater detention. All of these functions are indispensable in achieving habitat conditions needed by salmon for survival: well-moderated water temperature, clean channel substrate, structurally complex habitat with a variety of velocities, deep pools, cover and dissolved oxygen near saturation.

Undisturbed riparian and hillslope vegetation limit the frequency and magnitude of mass failures in headwater areas. The risk of initiating mass failures in headwater areas is increased by the combination of increases in subsurface water levels and decreases in root strength caused by the removal of vegetation. Field research has shown that roots help to stabilize granitic slopes by supplying cohesion (Burroughs and Thomas, 1977; Gray and Megahan, 1981). Groundwater levels exert a pronounced influence on the susceptibility of an area to mass failure (Gray and Megahan, 1983; Iverson and Major, 1986; Megahan and Bohn, 1989). Riparian vegetation reduces groundwater levels through evapotranspiration (Megahan, 1984b), by reducing rates of snowmelt discharge to the groundwater system (Gray and Megahan, 1981; Megahan, 1983), and by reducing the total snowmelt discharge through reduced snowpack accumulation (Male and Gray, 1981). The removal of riparian and hillslope vegetation increases groundwater levels by increasing snowpack accumulation, reducing evapotranspiration, and increasing snowmelt rates.

Even when debris flow failures are limited in both time and space, such failures dominate the long-term sediment budget in mountainous watersheds (Benda and Dunne, 1987). Based on surveys conducted in the late 1970s, it was estimated that the average volume of sediment annually delivered to the streams from landslides in the Idaho batholith was about 56,000 cubic yards; this rate of mass failure caused adverse fishery resource impacts throughout the study area (Megahan et al., 1978; Gray and Megahan, 1981).

Reductions in root strength and increases in riparian water table elevations following logging can also cause "sapping failures" in batholith soils. These failures differ from debris flows because they are progressive, smaller, and occur on gentle slopes (Megahan and Bohn, 1989). Although such failures are smaller, Megahan and Bohn (1989) estimated that during the year of maximum sapping erosion, sapping failures accounted for 2-56% of the post-disturbance sediment yields in three small watersheds in the Silver Creek watershed in Idaho.

The downstream impacts of debris flows on salmon habitat and survival are catastrophic and long lasting. Field studies have indicated that channel recovery may take 75 to 100 years (Kelsey, 1980). It has been well documented that salmon populations in the South Fork Salmon River have been significantly reduced by catastrophic sedimentation primarily caused by debris flows (Platts et al., 1989). Substrate conditions in the South Fork Salmon River still have not fully recovered from the mass failures which occurred in 1965 (Platts et al., 1989). Increased frequency of mass failures has the same effects on fish habitat conditions as increased surface erosion: increased fine sediment, decreased pool volumes, and channel widening.

Bank vegetation exerts a strong control on bank stability, fluvial channel erosion, and channel form (Graf, 1979; Richards, 1982; Ikeda and Izumi, 1990). Streamside vegetation provides bank stability through root strength (Graf, 1979; Richards, 1982). The loss of root strength can lead to bank collapse (Richards, 1982). Channel roughness reduces flow velocity which reduces the erosive power of streamflow. Riparian vegetation provides channel roughness in the form of LWD, roots, and standing stems. Increases in flow velocity increase the erosive power of streamflow that, in turn, typically increase channel erosion and downstream sediment transport (Richards, 1982). Gullying and stream incisement are often caused by the removal of riparian vegetation (Graf, 1979; Schumm et al., 1984; Harvey and Watson, 1986). Gullying and channel incisement can cause significant increases in sediment delivery (Schumm et al., 1984; MacDonald and Ritland, 1989). LWD supplied by riparian vegetation can reduce channel erosion by armoring channels (Sullivan et al., 1987).

Small perennial and ephemeral streams with high gradients and unconsolidated channel substrate in headwater areas are extremely sensitive to increased channel erosion (Rosgen, 1993). In areas where flows are dominated by snowmelt, these channel types are sensitive to increased peakflows caused by vegetation removal (King, 1989). Increased peakflows can lead to headward channel erosion or expansion of cross-sectional channel area (Megahan and Bohn, 1989; Heede, 1991; Gomez and Mullen, 1992). Increases in peakflow, alone, can increase erosion in smaller streams contributing to downstream sedimentation (Geppert et al., 1984; MacDonald and Ritland, 1989; King, 1989) in pools and low gradient stream reaches. The loss of bank stability and channel roughness in tandem with increased discharge renders these channel types extremely prone to increased channel erosion. Field reviews in the Blue Mountain Province have indicated that headward channel erosion commonly occurs in steep ephemeral channels in headwater systems in response to the loss of vegetation by fire or timber harvest, even when bank stability has not been lost (J. Rhodes, unpublished field notes, 1989 and 1993); ephemeral streams in Arizona were found to exhibit similar response to upstream timber harvest (Heede, 1991). However, headward gullying is most extreme in these stream types when bank stability and channel roughness elements have been lost due to vegetation removal. These small, headwater channels primarily transport only fine sediment due to low competency at shallow flow depths. Research on sediment transport in ephemeral channels in the Grande Ronde River watershed indicate that the vast bulk of sediment transported in the study streams is <0.08 in. in diameter (R. Gill, Wallowa-Whitman National Forest District Hydrologist, pers. comm., 1993). Once degraded, these high gradient streams in unconsolidated channel materials have very poor prospects for recovery, even after the environmental stresses have been eliminated (Rosgen, 1993).

Riparian vegetation also limits surface erosion near channels by overland flow or rainsplash. Ground cover reduces erosion and surface erosion increases as ground cover decreases (Dunne and Leopold, 1978; USFS, 1980; USFS, 1981). Small organic debris such as fallen needles and small branches from trees and shrubs, provide ground cover, in addition to forbs and grasses. Erosion near channels is of concern because the efficiency of transport of eroded soil to streams increases with proximity to the channel (Guy, 1970; NCASI, 1979; USFS, 1980).

The existence of overland flow increases both erosion and the efficiency of sediment transport of eroded sediment to streams. Near channel areas in floodplains and swale bottoms have the greatest duration and frequency of overland flow due to saturation of the soil profile (Dunne and Leopold, 1978; O'Loughlin, 1986). The duration and frequency of saturated soils in near channel areas is increased by the multiple effects of vegetation removal on water table elevation.

Hillslope and riparian vegetation provide filtering and storage of sediment before it can enter stream channels. This interception of eroded sediments reduces or delays the delivery of sediment to channels where it can be transported downstream (Heede et al., 1988). Many field studies have documented the effectiveness of riparian vegetation in arresting sediment that resulted from land disturbance such as logging (Murphy et al., 1986; Hartman et al., 1987; USGAO, 1988) and fire (Heede et al., 1988). The removal of riparian vegetation reduces the ability of the system to detain sediment from upstream and upslope sediment sources. Downed wood from riparian vegetation can provide significant sediment storage. For instance, about 1900 ft^3 of sediment can be stored behind a downed log with an average diameter of 3 feet and length of 100 feet that is lying parallel to slope contour on a hillslope with a gradient of 20%. After trees are removed, full recovery of natural levels of sediment storage behind downed logs in riparian zones probably takes about the same amount of time as in-channel LWD: 100 to 200 years depending on revegetation rates and the lifespan of trees.

During infrequent, but extreme, events, the sediment detention capacity of riparian systems can be completely overwhelmed even in old-growth areas, but especially in areas where riparian vegetation has been removed. Field inspection of the headwaters affected by the Tanner Fire and Flood Event of 1989 in the Upper Grande Ronde River on the Wallowa-Whitman National Forest, indicated that virtually all sediment storage was filled behind fallen old growth logs at distances greater than 100 feet from the stream channel; some of the fallen logs were greater than 3 feet in diameter at the base (J. Rhodes, unpublished field notes, 1990). The affected area was entirely covered by old growth prior to the fire. Despite the high levels of terrestrial sediment storage, significant amounts of sediment transported by flood flows were still swept downstream and deposited in salmon habitat. These observations indicate that vegetation loss within the area would have exacerbated downstream sediment loading to the Grande Ronde River. Sediment traps placed on side slopes at distances greater than 100 from the streams subsequent to the fire and flood events, were partially filled by sediment within a year (P. Boehne, Wallowa-Whitman National Forest Fish. Bio., pers. comm., 1993). These observations indicate that sediment filtering by riparian vegetation prevented significant amounts of sediment from entering the stream system during and after the event. These observations also indicate that sediment storage can be exhausted and overwhelmed during extreme events, even in old growth systems.

Large trees and downed wood in the riparian zone are effective in reducing both the distance that debris flows travel and the volume of sediment delivered to stream channels (Swanson et al., 1987). The effectiveness of riparian trees in partially arresting debris flows before entering streams has been repeatedly observed on the Clearwater National Forest in the Idaho batholith (R. Jones, Clearwater National Forest Hydrologist, pers. comm., 1991). Elsewhere, it has been documented that debris flows from clearcuts traveled about 150% farther than in forested areas (Ketcheson and Froelich, 1978).

Riparian vegetation also influences the rates of in-channel sediment routing to downstream salmon habitat by providing sediment storage behind LWD (Beschta, 1979; Megahan, 1982; MacDonald and Ritland, 1989). The amount of sediment stored behind in-channel LWD is significant. In a survey of sediment accumulations in streams in Idaho, Megahan (1982) found that LWD accounted for about 35% of the sediment storage sites and 49% of the stored sediment volume. Stored sediment volume in the channels was about 15 times the average annual sediment yield and was approximately equivalent to about 0.05 tons of sediment/ft of channel length (Megahan, 1982). If there is no continuing source of LWD recruitment to the channels, downstream sediment delivery will increase in two ways. First, as existing LWD decays without replacement, sediment currently stored behind LWD in the headwaters will be flushed downstream. For instance, Beschta (1979) found that the removal of LWD from an 820 ft section of a third order stream in the Oregon Cascades released approximately 6800 tons of stored sediment. Second, sediment delivered to the stream will also be flushed downstream rather than stored behind LWD obstructions if LWD sediment storage capacity is lost. If LWD does not exist to store and redistribute sediment, these sediments may become stored in pools or riffles. Swanson et al. (1987) noted that in-channel LWD can act as natural check dams for sediment and can reduce both the frequency and downstream mobility of debris flows. Field reviews on the Wallowa-Whitman National Forest and Clearwater National Forest have consistently indicated that LWD plays an integral role in sediment storage in steep headwater streams (J. Rhodes, unpublished field notes, 1989 and 1990).

Megahan (1982) observed that sediment storage sites provided by LWD lost about 97% of their effectiveness within six years of emplacement and concluded that retention of riparian trees was desirable from the perspective of helping to stabilize channel-sediment storage over time after timber harvest. Removal of trees within one old growth tree height of streams reduces LWD levels over time. However, LWD loss may also be accelerated by increases in channel width that render shorter LWD pieces unstable (Bisson et al., 1987).

Coniferous LWD provides more effective sediment storage than LWD from hardwoods in stream channels because of its greater lengths, greater diameters, and much greater resistance to decomposition (Sedell et al., 1988). Therefore, removal of conifers and/or conversion of riparian vegetation to hardwoods can reduce the effectiveness of LWD in storing sediment in-channel or on floodplains.

Riparian trees within one tree height of fish-bearing streams serve as a source of direct recruitment of LWD to fish habitat. LWD is lost over time as trees are removed within one tree height of streams or as tree heights are shifted towards smaller sizes that have lower recruitment rates to streams and are less stable in wide channels. Although most work to date has focused on the direct recruitment of LWD to stream channels from adjacent stands (McDade et al., 1990; Robison and Beschta, 1990) or downstream transport from upstream sources, it is also likely that stream channels annex the terrestrial legacy of downed wood in floodplains as the channels meander over time. Therefore, downed wood on floodplains provides an additional, indirect source of LWD.

LWD contributes to formation of large pools, and channel complexity. Salmon survival and production is positively correlated with both pool volume and LWD levels, though the LWD and pools typically exhibit covariance. LWD also provides habitat cover.

LWD impacts are especially serious because of the time required for full ecological recovery. On the west side of the Cascades, more than 200 years are needed for full ecological recovery after timber harvest (Gregory and Ashkenas, 1990). In the Snake River Basin, recovery of LWD may be somewhat less due to the generally shorter average lifespan of trees or longer due to slower regrowth; however, it is likely that the impacts from removing trees within one tree height will persist for more than 100 years.

Root anchoring from non-woody streamside vegetation also provides bank stability needed for the development of pools with overhanging banks. These habitats are important for both rearing and adult salmon (Platts, 1991).

Riparian vegetation moderates stream temperatures in both winter and summer in several ways. Shading of streamflow is crucial in controlling temperature increases (USFS, 1980; Theurer et al., 1985; Beschta et al., 1987). However, vegetation of a given height provides less stream shading as channel width increases (Theurer et al., 1984). In large, wide streams, shading from vegetation is limited and probably has a negligible effect on stream temperatures. However, riparian vegetation is critical to controlling channel width in alluvial channels (Ikeda and Izumi, 1990). Water temperatures at both the network and reach scale are extremely sensitive to channel width because channel width controls both the ability of vegetation to shade the stream and the surface area across which all energy exchanges occur (Theurer et al., 1984). Consequently, the role of riparian vegetation in controlling channel width may be more important in wider streams than from shading alone.

Shading by riparian vegetation on small, headwater streams exerts a powerful control on water temperatures at both the network and reach scale. At the reach scale, riparian vegetation can almost completely shade the stream surface of smaller streams. Flows in small headwater streams tend to be shallow and sluggish during the summer which render them very susceptible to solar heating (USFS, 1980). At the network scale, shading on small streams is important to water temperatures because the smaller streams comprise the majority of the channel network by length and exert a strong effect on the downstream temperature profile. In western Oregon, full retention of all vegetation within about 100 feet of streams is required to maintain natural shade conditions (Beschta et al., 1987), although a variety of factors influence vegetative shading including slope, aspect, vegetation height, and orientation.

Riparian vegetation also maintains water temperatures by influencing near-stream air temperatures. Although solar radiation is the dominant heat source to streams during the summer in the western U.S., convection is typically the second largest source of heating (Male and Gray, 1981). Air temperature at the stream surface is a dominant control on heat transfer via convection and conduction (Campbell, 1977; Male and Gray, 1981). When air temperatures are greater than stream temperature, convection and conduction heat streams. The rate of stream heating from convection and conduction increase as the temperature differential between the air and the stream increase, other factors remaining equal. The air temperature profile above a stream also is one of the dominant controls on equilibrium temperature. Equilibrium temperature is the temperature that still water eventually reaches under steady heat fluxes and steady meteorological conditions. Although streams typically do not reach equilibrium temperatures, equilibrium temperatures limit the ultimate downstream maxima in wide stream reaches far from the channel divide where the influence of shade and upstream water temperatures has diminished. The loss of riparian vegetation increases midday air temperatures near the stream during the summer. The loss of vegetation outside of the riparian zones can also increase air temperatures near the stream, when the vegetation loss is near enough to allow the penetration of meteorological conditions in the disturbed area into the area near the stream (Fritschen, 1970 as cited in Harris, 1984; Franklin and Forman, 1987; Chen, 1991 as cited in USFS et al., 1993). Limited research on hillslopes in western Washington indicates that clearcuts increase air temperatures in undisturbed forests for a distance of about 2.25 tree heights (Chen, 1991 as cited in USFS et al., 1993). However, the distance that meteorological conditions in disturbed areas penetrate into undisturbed forest stands is probably a function of an array of factors. With other environmental variables constant, penetration distance of higher summer air temperatures from cutover areas into undisturbed forest stands is probably a function of vegetation density and vegetation height because these factors control the flux of moisture, solar radiation, and air into and out of an undisturbed stand (Campbell, 1977). Therefore, the distance that air temperatures are increased in undisturbed riparian zones by adjacent disturbance could be greater in the Snake River Basin than west of the Cascades because of generally lower tree density and tree height. While the data of Chen (1991 as cited in USFS et al., 1993), may not be completely applicable to stream conditions in the Snake River Basin, the effect of undisturbed riparian vegetation on air temperatures over streams is probably a critically important concern. Ambient summer air temperatures outside of forests can approach or exceed 100°F in parts of the Snake River Basin. Therefore, increases in air temperatures over streams caused by vegetation loss have considerable potential for increasing water temperatures in salmon habitats at the reach and network scale.

In summer, the elevation of water temperatures can kill juveniles outright, drive them out of a stream segment (Lindsay et al., 1986), reduce growth (Theurer et al., 1985; Armour, 1991), increase the incidence and virulence of disease (Theurer et al., 1985; Fryer and Pilcher, 1974; Fryer et al., 1976; Groberg et al., 1978), and reduce the amount of available rearing habitat (Theurer et al., 1985). On a system-wide basis, maintenance and restoration of the maximum amount of shade and riparian vegetation ecologically sustainable on a long-term basis would result in the greatest areal extent of suitable summer habitat for coldwater dependent species. In winter, low water temperatures are exacerbated by canopy removal (Platts, 1984) or increases in channel width. Reduced winter water temperatures reduce growth of juveniles and can also cause the formation of anchor ice that smothers all aquatic life in the channel bed (Platts, 1984).

The avoidance of water temperature increases is critical because of the time required for ecological recovery. In the western Cascades, riparian shading and water temperatures require about 25 years for ecological recovery in small streams (Gregory and Ashkenas, 1990), although the recovery time varies with elevation. Small streams in higher elevation sites in the western Cascades may require more than 40 years for the ecological recovery of shading and water temperature after vegetation loss due to slower rates of vegetative regrowth (Beschta et al., 1987). Stream size also affects the time needed for recovery of shading and water temperature. The larger the stream, the greater the time needed for restoration of effective shading after disturbance. Large streams require tall trees for shade protection and probably much longer periods than small streams for ecological recovery of shading and water temperature after disturbance (Beschta et al., 1987). Recovery times for shading in Snake River Basin may be longer than those found in the western Cascades due to generally slower rates of revegetation and regrowth.

Riparian areas also provide important hydrologic functions. Floodplains store and detain water during flood events. In some forested mountainous watersheds with snow dominated hydrology, almost all baseflow is transmitted through riparian soils, especially those in meadow systems (Rhodes, 1985). Riparian areas also store streamflow during high flow events as bank storage. As stream stage drops lower than adjacent groundwater levels, riparian areas release water stored as bank storage. Compaction of riparian soils reduces the amount of available water storage within the riparian profile. Compaction may also increase the amount, frequency and duration of overland flow near streams by decreasing soil conductivity, soil porosity, (Gardner and Chong, 1990; Purser and Cundy, 1992) and infiltration rates (Gardner and Chong, 1990). Increases in overland flow and decreases in water storage near streams can translate into lost recharge of groundwater in riparian zones, and, ultimately, reduced baseflow inputs during low flow periods.

The loss of riparian vegetation may also reduce baseflows in some stream systems (Ponce and Lindquist, 1990). Activities that remove vegetation can compact soils and reduce infiltration rates, causing reductions in recharge and baseflow. Vegetation loss and associated bank disturbance can also lead to considerable channel incision in some stream types. Channel incision vertically de-links shallow riparian groundwater systems from streams, reducing recharge to riparian zones from streamflow. Where stream incision is severe, the channels only serve as groundwater drains, reducing local groundwater elevation and baseflow over time.

Reductions in baseflow cause increased seasonal temperature extremes, because groundwater warms streams in winter and cools them in the summer. Seasonal temperature extremes are also exacerbated by flow reductions because streams heat and cool faster at lower flows under a given energy budget. Reductions in baseflows also reduce pool depth and the amount of usable habitat. Complex side channel habitat may become inaccessible. In some cases, flow reduction may also reduce habitat cover, because cover tends to be greatest along channel margins.

3.1.2 Activities affecting riparian zones: Grazing, road construction, mining, and logging all affect riparian zones directly and indirectly. Grazing has profound effects on riparian zones because grazing intensity is typically greatest in riparian areas due to thermal cover and the availability of water and forage. Through a variety of mechanisms, grazing negatively affects several aspects of riparian vegetation. Grazing can lead to the alteration of the composition of vegetative communities, loss of deep rooting plant species, decreased canopy cover, and decreases in size of vegetation (Platts, 1991). Due to the combined effects of grazing on soils, bank vegetation, bank stability, and resultant channel morphology, grazing can reduce water table elevations leading to shifts in plant communities and shade levels (Platts, 1991). Bank trampling and churning by livestock take a major toll on both bank vegetation and channel morphology. On Bear Valley Creek, bank stability appears to be inversely correlated to grazing use, as measured by forage utilization (Boise National Forest, 1993). Grazing also compacts soils, reducing infiltration rates, hydraulic conductivity, and available water storage within soil layers. Gifford and Hawkins (1978) concluded that grazing at any intensity decreased infiltration; "moderate" and "light" intensity grazing did not have statistically detectable different effects on infiltration. These alterations in soil properties can lead to reduced groundwater levels and increased frequency and duration of overland flow.

Grazing also increases erosion by reducing ground and shrub cover. Schulz and Leininger (1990) found that shrub and litter cover were respectively, 2 and 5.5 times higher within exclosures than within grazed plots; grazed areas had four times the area of bare ground than within exclosures.

The same effects of grazing that degrade riparian systems, also prevent the recovery of riparian vegetation, channel morphology, and hydrologic function. While some effects of grazing may be ameliorated by some grazing strategies, field reviews have consistently indicated that there is little chance of recovery for deep rooted riparian vegetation in systems with non-cohesive soils where the trampling and churning of streambanks continues. Available information indicates that grazing generally retards the establishment and growth of deep-rooted riparian species that provide shade (Green, 1991; J. Kauffman, Ore. State Univ. Prof. of Rangeland Resources, pers. comm., 1992) and bank stability (Kauffman et al., 1983; Boise National Forest, 1993 (See Figure 32)). In degraded riparian areas, there is low probability of successful vegetative recovery without some period of rest from livestock pressure. The suspension of riparian grazing, whether temporary or long-term, is the grazing management strategy most compatible with vegetative recovery and has the lowest risk of failure (Clary and Webster, 1991; Platts, 1991; Beschta et al., 1991; Elmore, 1992; Anderson et al., 1993; Kauffman et al., 1993).

Grazing can also disrupt the influence of riparian zones on streams by re-routing streams. Livestock trails to streams in degraded systems sometimes pirate streams away from established channels and riparian vegetation (J. Rhodes, unpublished field notes, 1990).

Although little research has been done on the effect of grazing on coniferous vegetation and trees, it appears that grazing has a limited effect on the size and stocking levels of coniferous vegetation. However, grazing may reduce coniferous regrowth through the trampling of seedlings. Grazing may also promote the invasion of formerly wet meadows by non-phreatic conifers, via lowering of the water table. A positive feedback loop exists between vegetation loss and grazing use. Livestock use becomes heavier in riparian zones as vegetation density is reduced because livestock access is facilitated.

Logging within riparian areas removes coniferous vegetation and compacts soils. Yarding also removes ground vegetation and ground cover. Yarding and site preparation can cause additional losses of downed material, ground cover, and riparian vegetation. Logging in or near riparian zones can indirectly affect riparian zones by increasing the frequency of blowdown along the edges of roads and clearcuts (Franklin and Forman, 1987). Although the findings of Franklin and Forman pertain to forests west of the Cascades and were not specific to riparian areas, it is likely that the results are somewhat applicable to riparian zones in the Snake River Basin.

Riparian vegetation can also be disrupted by mass failures. Where the failures hit riparian systems, they can knock down trees, or cause tree mortality through root burial. Mass failures can completely eliminate riparian vegetation within affected reaches (Furniss et al., 1991). Most studies have indicated that logging increases the frequency and volume of mass failures relative to undisturbed areas (Dunne and Leopold, 1978; Furniss et al., 1991).

Road construction within riparian areas removes all vegetation from the road prism and severely compacts soils. The duration of the effects of roads on riparian vegetation is far greater than logging, because roads prevent vegetative recovery for the life of the road. The frequency and magnitude of mass failures from roads is higher than from logging per unit area (Dunne and Leopold, 1978; Megahan et al., 1978; Geppert et al., 1984; Furniss et al., 1991). Streams are also often re-routed from established channels and riparian zones to make way for roads. Roads may also alter water table height in riparian systems by intercepting subsurface flow on hillslopes (Megahan, 1972) or reducing subsurface flow into riparian zones through the effects of soil compaction. Megahan (1972) speculated that subsurface flow interception by roads could affect downslope vegetation by reducing soil moisture. Developed recreation probably has the same effects as roads on riparian functions.

Surface mining in riparian zones severely disrupts all riparian functions (Nelson et al., 1991). Surface mining removes vegetation and soils (Nelson et al., 1991). Mining disrupts surface and subsurface flow routing via alteration of both soils and topography (Nelson et al., 1991). Dredge mining completely alters channel form, vegetation, and the composition of surface soils (Nelson et al., 1991). Streams are re-routed by dredge spoils or moved to facilitate mining. Due to the alteration of soil conditions, revegetation and recovery in mined areas can be extremely slow (Nelson et al., 1991). Water pollution caused by mining can kill riparian vegetation and preclude revegetation.

3.1.3 Evaluation: Fully functioning riparian areas are clearly vital to salmon survival. The protection of riparian areas is a necessary, but not sufficient, condition for habitat protection and recovery. Full protection of riparian zones, alone, may not fully protect salmon habitat from disruption in ecological functions at the watershed scale. Habitat conditions amenable to salmon survival cannot be achieved by protecting only riparian zones along fish-bearing reaches; tributary channels also need protection if downstream salmon habitats are to be protected and improved. Ephemeral and perennial streams without salmon typically comprise 60-90% of the channel network by length in watersheds with salmon. These non-fish-bearing streams convey a considerable portion of the annual flow and exert a tremendous influence on sediment transport, channel structure, seasonal flow volumes, and water temperatures in downstream fish habitat. In most managed systems in the Snake River Basin, riparian areas have been considerably damaged by mining, grazing, logging, and/or road construction. Full protection of riparian systems from further damage is a critical need, if salmon habitat is to improve.

Riparian zone functions that reduce sediment delivery are critical because erosion and sediment delivery in most managed systems have been greatly increased by land use activities. Grazing is widespread in the Snake River Basin and has greatly increased erosion and sediment delivery in systems such as the Upper Grande Ronde River (Beschta et al., 1991; Anderson et al., 1992) the Imnaha headwaters (J. Rhodes, unpublished field notes, 1989 and 1993), and Salmon River tributaries, such as Johnson and Bear Valley Creeks (Beschta et al., 1993; Boise National Forest, 1993; NMFS, 1993). In the Upper Grande Ronde River, Bear Valley Creek, and Johnson Creek, increases in sediment delivery from grazing have contributed to levels of fine sediment that have greatly diminished salmon survival (Anderson et al., 1992; Boise National Forest, 1993; NMFS, 1993), channel widening, and substantial losses in pools (McIntosh, 1992; Boise National Forest, 1993; McIntosh et al., 1994). In these same watersheds, mining has also significantly contributed to the disruption of riparian zones and increased sediment loading (Anderson et al., 1992; McIntosh, 1992; Boise National Forest, 1993).

Loss of root strength in small channel corridors, banks and/or headwalls greatly contributes to mass erosion in the Idaho batholith (Megahan et al., 1978; Gray and Megahan, 1981; Megahan and Bohn, 1989). Mass failures from roads were the dominant cause of the catastrophic sedimentation of the South Fork Salmon River that contributed to the precipitous declines in chinook salmon in habitat that was once the major producer of salmon in the Salmon River (Platts et al., 1989). In a survey of more than 1,400 landslides on the Boise and Clearwater National Forests (Megahan et al., 1978), roads were overwhelmingly associated with the mass failures. Roads, alone, accounted for 58% of the failures, while roads in combination with logging and/or fire accounted for 88% of the failures, while only 3% occurred in undisturbed settings (Megahan, et al., 1978; Gray and Megahan, 1981). Sapping failure in small channels can be a significant source of elevated sediment delivery in the Idaho batholith (Megahan and Bohn, 1989).

Many riparian corridors in the Snake River Basin are occupied by roads. Field surveys in the Imnaha, Grande Ronde, Salmon, and Clearwater drainages have consistently indicated that surface erosion from roads in riparian areas delivers substantial amounts of sediment to stream systems.

Logging has also elevated sediment delivery considerably, throughout the Snake River Basin. Available data and field reviews indicate that in many watersheds, logging and road construction have been disproportionally concentrated in riparian zones (See Figure 33). Besides elevating on-site erosion, logging and road construction in the Blue Mountain Province have also contributed to increased headward channel erosion (J. Rhodes, unpublished field notes, 1989 and 1993). However, headward channel erosion has been greatest under the combined impacts of logging and grazing.

Protection of riparian vegetation by avoiding disturbance in riparian zones have repeatedly been cited as one of the most effective approaches to limiting sedimentation caused by land disturbance (Furniss et al., 1991). In their study of the effect of forest vegetation removal on slope stability, Gray and Megahan (1981) concluded that in order to reduce mass erosion hazards from clearcuts and roads, "Leave buffer zone of trees above and below haul roads...Leave buffer zone of undisturbed vegetation along **all** streams" (emphasis added). Heede et al. (1988) concluded, "Implications of this study are that land managers, concerned to avoid erosion and sedimentation following disturbance should concentrate on the establishment and enhancement of vegetation buffer strips along channel banks." In an overview of watershed impacts from logging roads, Megahan (1984a) concluded that "providing a maximum of obstructions to catch and retain sediment before it reaches the drainage system..." in concert with hazard avoidance is "...usually, by far, the most efficient and cost effective means to reduce downstream sedimentation impacts."

Although undisturbed riparian vegetation can be effective in reducing sediment delivery to stream channels, it is unlikely that it can completely eliminate accelerated sediment delivery to stream systems caused by upslope land disturbances. Available research indicates that logging activities still elevate sediment delivery to streams at increased rates even with "no-cut" buffers. Megahan (1987) found that sediment delivery from helicopter logging and prescribed burning in the Idaho batholith increased sediment delivery by more than 100% in a watershed where 75 foot buffers were provided. Despite the buffers, the statistically significant increases in sediment delivery persisted for more than 9 years. Channel measurements indicated that channel erosion was <u>not</u> the cause of the increased sediment delivery. Heede (1991) found that increased peakflow caused by logging and road construction in the mountains of Arizona increased channel erosion in ephemeral channels that were protected by buffers of about 150 feet of undisturbed area. In Pennsylvania, Lynch and Corbett (1990) found that turbidity and suspended sediment were increased in logged watersheds where all perennial streams were provided with 100 foot, no-cut buffers; the increased sediment was partially attributed to blowdown of portions of the buffers. Streamside buffers of undisturbed vegetation may not be a panacea for sediment delivery from logging to stream channels. Although riparian reserves can limit accelerated sediment delivery from grazing, logging, roading, and mining, they probably cannot eliminate increased sediment loading to streams caused by land disturbance.

The effects of removing riparian vegetation on non-fish-bearing streams has significant effects on downstream fish habitat due to their inherent sensitivity to disturbance, hydrologic linkage to downstream habitat conditions, and the fraction of the channel system comprised by smaller ephemeral and perennial streams. Increased channel erosion in small headwater streams is likely to primarily transport fine sediment that is deleterious to salmon survival.

The removal and loss of riparian vegetation has elevated water temperatures considerably throughout the Snake River Basin. Shade data are limited, but indicate that grazed watersheds with logging generally have low levels of shading and high water temperatures. In the Upper Grande Ronde, it is estimated that stream shading is currently at about 28%; potential stream shading is estimated to be about 72% (Anderson et al., 1992). Water temperatures in the Grande Ronde River now regularly exceed levels that are directly lethal to salmon (Anderson et al., 1993). Winter icing is also a problem in the Upper Grande Ronde. The loss of shading in the Upper Grande Ronde has been caused by the combined effects of logging, road construction, mining, and grazing.

On the Clearwater National Forest, water temperatures in excess of 68^{o}F occur regularly during the summer on Eldorado and Lolo creeks (Nez Perce Tribe, unpublished data). Both Lolo and Eldorado creeks have undergone extensive logging and road construction in the riparian areas; it is estimated that about 25% of the riparian zone in Eldorado has been logged or roaded over the past

30 years (Clearwater National Forest, 1992).

On the Tucannon River, the elevation of water temperatures caused by the loss of riparian vegetation is estimated to have rendered about 24 miles of mainstem habitat unusable by salmon (Theurer et al., 1985 (See Figure 25). Theurer et al. (1985) concluded, "The change in temperature regime caused by the loss of riparian vegetation alone is sufficient to explain the reduction in the salmonid population in the Tucannon River." The primary cause of water temperature elevation was attributed to vegetation loss resulting in reduced shading and channel widening (Theurer et al., 1985). Loss of riparian vegetation was caused by a combination of logging, grazing, roads, and agriculture on private lands. Logging and roading have been considerable in riparian zones in some small subcatchments within the Tucannon River watershed (Umatilla National Forest, unpublished data (See Figure 33)).

Roads and logging in riparian zones have also reduced LWD levels throughout many systems in the Snake River Basin. Many watersheds in the Blue Mountain Province with an extensive history of logging have streams with low levels of LWD (See Figures 20 and 34). Data are limited on the condition of potential sources of LWD, such as riparian stocking levels. However, available data indicate that watersheds managed for timber production have diminished the areas of riparian zones with late seral state vegetation (Henjum et al., 1994). Pool losses in the Grande Ronde system (See Figure 10) have been attributed to the loss in LWD over time (McIntosh, 1992).

Field reviews consistently indicate that channel incisement in wet meadows is relatively common in grazed areas in the Blue Mountain Province and the Idaho batholith (Beschta et al., 1991; M. Purser, Conf. Tribes of the Umatilla Indian Reservation Hydrologist, pers. comm., 1992; Beschta et al., 1993). It is likely that these conditions have reduced local water table elevations and baseflow contributions in the affected reaches during the summer (Beschta et al., 1993). However, increased baseflow caused by the effects of vegetation removal and tree mortality from disease at the watershed scale may have offset losses due to channel incision and meadow desiccation, because data indicate that summer flows in the Grande Ronde River have increased over the past 50 years (McIntosh, 1992).

Riparian reserves must be fully functional to be most effective in protecting habitat and ameliorating the effects of basin level disturbance. However, most riparian systems in managed basins are not fully functional. A considerable amount of riparian zones in managed watersheds have been disturbed by logging, roading, grazing, and/or mining. It is difficult to estimate the amount of riparian zones that have been affected by the various activities because monitoring of riparian conditions is lacking. Nonetheless, some available information can be used to estimate the amount of riparian disturbance caused by logging. Until the last few years, riparian areas were generally harvested at the same rate as all other lands. Although the relationship between the amount of logging-related disturbance at the watershed level to that incurred in riparian zones probably varies among watersheds, watershed level estimates of disturbance provide an index of the amount of riparian zones that have already been disturbed. In the Upper Grande Ronde River, 17 to 42% of the area of subwatersheds have been logged or roaded over the past 30 years (Wallowa-Whitman National Forest, unpublished data). Therefore, about 17 to 42% of the entire riparian areas of these watersheds may have been logged off. The WWNF estimates that over the last decade, riparian areas were logged at a rate of about 1,200 acres/yr (Wallowa-Whitman National Forest, 1990). It is estimated from data in the Umatilla National Forest Plan (Umatilla National Forest, 1990) that it harvested about 25% of the riparian areas on the Umatilla National Forest over the past 30 years. It can be assumed that riparian systems disturbed by logging and roads have reduced shading, increased vulnerability to windthrow, higher summer air temperatures, reduced rates of LWD input, elevated sediment delivery, reduced sediment detention capacity, reduced bank stability, and compacted soils, although the magnitude of these perturbations will vary by site and treatment.

Data are generally not available to estimate the amount of riparian zones that have been degraded by grazing and mining, and existing range assessments are, unfortunately, irrelevant to assessing riparian/habitat conditions. However, most of the national forests in the Snake Basin support major grazing programs; grazing is typically heaviest in riparian systems. Field reviews and site-specific assessments indicate that it is likely that the majority of riparian zones within grazed areas have been degraded to some extent. It can be assumed that riparian systems disturbed by grazing have reduced shading, higher summer air temperatures, elevated erosion rates, reduced sediment detention capacity, reduced bank stability, increased channel erosion, and compacted soils, although the magnitude of these perturbations will vary by site and grazing history. It is, therefore, clear that most riparian systems are not fully functional, and this must be factored into recommendations for the width of riparian reserves.

Riparian reserves have been a key component of recovery strategies aimed at protecting salmon and salmon habitat (Anderson et al., 1992; USFS, 1993b; USFS et al., 1993; Henjum et al., 1994). This consensus among teams from a variety of disciplines and affiliations, provides compelling evidence of the need and importance of riparian reserves. However, the recommended size of riparian reserves varies, and so do the activities allowed within the reserves (Anderson et al., 1992; USFS, 1993b; USFS et al., 1993; Henjum et al., 1994).

Our recommendations regarding the widths of riparian reserves are based on several considerations. First, reserve widths must be adequate to fully protect all identifiable riparian functions, based on the best available information. Any reduction in the function of riparian zones has the potential to cause degradation of salmon habitat over time and forecloses management options for the protection and improvement of habitat conditions. Figure 35 displays generalized widths needed <u>fully</u> protect the ecological functions of riparian zones, based on best available information. Riparian functions that depend on old growth forest characteristics require more than 200 years for ecological recovery after disturbance. Reserve widths should be set based on conditions that are relatively likely to occur over 100 to 200 years, such as change in channel position in floodplains or large storm events. Floodplains are the temporal extension of streams. Because streams meander across floodplains, as well as occupying them during flood events, floodplains should be given the same protection as active stream channels so that adequate riparian conditions exist during flood events and as channels shift over time. Likewise, riparian function during extreme, but infrequent, events such as flooding, fire, and/or landsliding must also be factored into width considerations given the longevity of potential impacts on riparian function from vegetation removal. Smaller channels that do not provide habitat for fish should receive at least as much protection as larger salmon-bearing streams. Smaller streams comprise the bulk of the channel network, exert a strong control on downstream habitat conditions, and are extremely sensitive degradation caused by vegetation loss. Degradation of non-fish-bearing streams will propagate downstream and cause habitat degradation in salmon habitats. Most riparian systems have been perturbed and currently are not fully functional. Environmental brinkmanship is undesirable in developing protection measures because the effects of inadequate protection may not be reversible. Therefore, for the foreseeable future, factors of safety should be included in developing recommended widths for reserves, just as safety factors are included in bridge design.

We recommend that riparian reserves extend **a minimum** of 300 feet slope distance from each side of floodplains, or to the top of a topographic divide, whichever is less. The reserves apply to all streams. Within these riparian reserves, no additional anthropogenic disturbance of soils and vegetation should occur until it has been documented that the habitat conditions in most (>90%) managed salmon habitats have improved, or that salmon populations have recovered (See Section 3.7 Geographic Criteria for Re-evaluating Land Use Standards). Where habitat standards are not met, the active restoration or removal of existing disturbances within the reserves (such as surface mining, developed recreation sites, roads etc.) should be undertaken.

Reserves 300 feet from the edge of floodplains should be effective in protecting stream and floodplain shading, LWD loading to streams and floodplains, limiting additional sources of sediment delivery close to streams, increasing bank stability and root strength over time, and maintaining and restoring ability of riparian reserves to filter and detain sediment. The recommended width may not fully protect riparian vegetation from accelerated windthrow over time due to edge effects. The recommended width may also be inadequate to buffer streams against high levels of sediment delivery and/or increased peakflows caused by upslope land disturbance, especially during extreme events. Although the riparian reserve widths should adequately protect floodplain soils and local controls on subsurface hydrology, the reserves do not ensure that subsurface hydrology and baseflow mechanisms are fully protected because roads and other activities outside the reserves can disrupt subsurface hydrology. Disruptions in subsurface hydrology can increase summer water temperatures by directly heating water brought to the surface or by reducing summer streamflow. Therefore, the riparian reserves, alone, may not be adequate to completely protect against increases in summer water temperatures. However, **fully functional** riparian reserves of 300 feet from the outer edge of floodplains provide a high likelihood of preventing degradation and facilitating improvement in salmon habitat conditions and salmon survival.

Unfortunately, many riparian systems are not fully functional and impacts to stream and riparian systems will continue to accrue. Until these sources of degradation within the reserves are removed or adequately addressed, the effectiveness of the riparian reserves in protecting and restoring salmon habitat will be limited; in some cases, it will be unsuccessful. However, in the absence of riparian reserves, it is unlikely that salmon habitat can be protected and restored over time, except in roadless and wilderness areas. Although the expansion of reserve widths can reduce risk to fish habitat, it cannot offset continuing habitat damage caused from impacts within the reserves. Therefore, we recommend that passive and active restoration efforts should focus on eliminating any persistent effects from existing impacts, such as mining, within the reserves, especially when habitat standards are not met, rather than attempting to offset on-going impacts through expansion of the reserves beyond 300 ft.

We do not recommend the use of approaches to riparian restoration that involve vegetation removal. Such approaches are fraught with risks of disrupting various riparian functions such as shading and hydrologic routing, and their effectiveness in improving riparian functions and habitat conditions remains a matter of heuristic speculation. Silvicultural approaches involving vegetation removal in riparian reserves should not be considered until the approaches have been documented to have been successful under ecologically applicable experimental conditions, and that habitat and riparian conditions have improved in the majority of Snake Basin watersheds that provide salmon habitat. Riparian restoration efforts should focus on activities that are low risk and likely to be effective, such as suspension of grazing in degraded reaches.

Suspension of riparian grazing in degraded reaches (or in degraded watersheds) is probably the most effective approach to restoring riparian systems and realizing rapid habitat improvement in the Snake River Basin. Livestock grazing in watersheds where water temperature standards are not met in salmon habitat should be suspended within the riparian reserves until water temperature standards are met, or a statistically significant ($p<0.05$) improving trend over at least 5 years is documented through monitoring. In watersheds where bank stability standards are not met, we recommend that grazing be suspended within a distance of half of a tree height from the edge of the floodplain or from the edge of the stream where floodplains are absent, until the bank stability standard is met, or monitoring documents that a statistically significant ($p<0.05$) improving trend has occurred over at least five years. Notably, other evaluations of habitat conditions and protection measures have also recommended the temporary or permanent elimination of riparian grazing in portions of the Snake Basin in order to protect and restore conditions in degraded salmon habitat (Beschta et al., 1991; Anderson et al., 1993; Beschta et al., 1993; Henjum et al., 1994).

In non-degraded riparian areas along reaches and in watersheds where habitat standards are met, grazing should be suspended until allotment management plans are revised to be as compatible as possible with protection of vegetation, channels, and soils. All grazing should be closely monitored. We strongly recommend that all grazing units should have several riparian exclosures in representative areas for use as a monitoring references. Where trends in bank, vegetation, or habitat conditions in grazed areas are worse than trends within the exclosures, grazing should be suspended until trends stabilize and a more compatible grazing strategy can be found.

We also recommend that data on riparian disturbances and conditions within watersheds be required. At a minimum, data should include the number of road crossings, the road mileage within riparian reserve width, area logged within the reserves by harvest method, amount and type of mining within reserves, grazed area within reserves, and stream shading. Until these data are collected, on-going or existing activities that disturb soils or vegetation within the reserves should be suspended.

3.2 SEDIMENT DELIVERY

3.2.1 Effects on salmon: Sediment delivery has profound impacts on salmon and salmon habitat because it is one of the primary stream processes that control stream channel conditions at scales ranging from the reach to the network (Schumm, 1969, Richards, 1982; Lisle, 1982; Carson and Griffiths, 1987; Dietrich et al., 1989; Rosgen, 1993). There are several factors that account for why accelerated sediment delivery so profoundly affects the survival and production of salmonids, and especially chinook salmon. First, chinook salmon spawn and rear in depositional stream environments that are sensitive to sedimentation due to reach properties (gradient) and position within the watershed (Frissell, 1992; Rosgen, 1993). Second, shifts in sediment delivery have multiple effects on channel conditions and processes. Third, habitat changes caused by accelerated sediment delivery have density-independent effects that lower salmon survival even at low seeding levels. Fourth, the changes in channel conditions caused by changes in sediment delivery not only have multiple effects on salmon at several lifestages, but also affect the food web, and water quality conditions, such as water temperature and dissolved oxygen. These combined changes can work synergistically to reduce salmon survival. Fourth, activities that degrade other aspects of habitat conditions, such as LWD or stream shading, also typically elevate sediment delivery. In their review of the effect of sediment on aquatic life, Cordone and Kelly (1961) stated that accelerated erosion was probably the most "insidious" of the factors causing the loss of fishery resources.

Shifts in the magnitude, timing, or composition of sediment delivery to streams do not have a single, simple effect on channel conditions. Rather, shifts in sediment loads set off a complex of channel responses including changes in channel gradient, sinuosity, pool volumes, pool frequency, channel width, channel cross-sections, channel network geometry, particle size distribution in stream substrate, bedload transport, and suspended sediment loads. Although the qualitative response of channel conditions to shifts in sediment delivery have been widely studied and are generally well-understood (Schumm, 1969; Richards, 1982; Lisle, 1982; Carson and Griffiths, 1987; Rosgen, 1993), the magnitude of change in channel conditions is difficult to predict in field situations with accuracy. The magnitude, duration, and extent of channel response varies considerably with respect to geomorphology, geology, hydrology, vegetation, climate, and initial channel conditions, as well as the characteristics of the shift in sediment delivery. Other factors remaining equal, the likelihood of deleterious channel change is strongly dependent on the magnitude of the increase in sediment delivery relative to the natural sediment delivery, because channels have adjusted to natural sediment delivery rates over time (Richards, 1982). A qualitative overview of channel response to changes in discharge and bedload transport is shown in Table 2 (Schumm, 1969). Only a few of the linkages will be discussed because a full and complete review of all factors mediating channel metamorphosis in response to increased sediment delivery is beyond the scope of this report. For more detail, the reader is referred to the citations. Additional detail can also be found in the sections on channel substrate (1.1), pools and LWD (1.2.1), riparian reserves (3.1), sediment delivery (3.3), and logging-related disturbance (3.4).

Field, flume, and theoretical investigations show that increased sediment delivery reduces pool volumes (Lisle, 1982; Lyons and Beschta, 1983; Jackson and Beschta, 1984; Lisle and Hilton, 1992), increases fine sediment levels (USFS, 1983; MacDonald and Ritland, 1989; Dietrich et al., 1989; Platts et al., 1991; Diplas, 1991; Lisle and Hilton, 1992), increases the frequency and duration of scour and fill events (Howard, 1987; Lisle and Hilton, 1992); increases suspended sediment and turbidity (Guy, 1970; Geppert et al., 1984; Anderson and Potts, 1987), and increases channel width (Schumm, 1969; Jackson and Beschta, 1984; Carson and Griffiths, 1987; Lisle and Hilton, 1992) which can intensify seasonal water temperature extremes (Theurer et al., 1984; Alexander and Hansen, 1986). Separately these changes in channel conditions reduce the ability of habitats to produce salmon and, in some cases, elevate salmon mortality. However, sediment delivery has combined effects on salmon because the changes in salmon habitat caused by increased sediment delivery do not occur separately, but are part of a complex of processes involved in channel adjustment to increased sediment delivery.

A wide variety of data indicate that increases in fine sediment reduce salmon STE (Everest et al., 1985; Everest et al., 1987; Scully and Petrosky, 1991) by entombment (Chapman and McLeod, 1987) and reductions in levels of intergravel dissolved oxygen during the incubation period (Maret et al., 1993). As discussed in Section 1.1 Channel Substrate, it is likely that the sedimentation of redds is rapid after spawning in streams with high sediment loads and/or high levels of fine sediment upstream of the redds. Shifts towards reduced particle sizes in redds also render them more susceptible to scour, because smaller particles can be moved at lower flows. Changes in channel morphology caused by high sediment loads tend to increase the frequency and duration of fine sediment transport and, thus, increase the propensity for sedimentation of redds by fine sediment subsequent to spawning.

Changes in channel morphology in response to elevated sediment delivery tend to increase the frequency and duration of sediment transport, especially during lower flow periods even in the absence of shifts towards smaller channel substrate. High levels of sediment delivery typically cause channel widening, a steepening of gradient, and reductions in channel depth (Schumm, 1969; Jackson and Beschta, 1984 (See Table 2)). The changes in channel morphology tend to increase the ability of a stream to transport sediment transport at a given discharge (Lisle, 1982; Lisle and Hilton, 1992).

High levels of sediment delivery also lead to pool loss. Pool loss may increase the probability of sedimentation of salmon redds because pools can act as traps for fine sediment in transport. During periods of sediment transport at lower flows, fine sediment is deposited in pools (Lisle and Hilton, 1992), which may prevent or reduce downstream sediment transport and/or sedimentation (Alexander and Hansen, 1983). As pools are lost or reduced in size upstream of spawning reaches, the trapping of sediment in transport prior to reaching the spawning and rearing habitat is reduced. Thus, the effect of increased sediment delivery on channel morphology increases the likelihood of sedimentation within redds leading to reduced salmon STE.

Loss of pools and pool volumes caused by accelerated sediment delivery also negatively affects rearing and adult salmon. Pools are used for rearing by juveniles, resting by migrating adults, and as refugia during droughts and floods.

Increased sediment delivery also reduces available rearing habitat by reducing interstitial spaces between cobbles (increasing cobble embeddedness) via the same mechanisms that increase fine sediment levels and sedimentation of redds. Increased cobble embeddedness in salmon rearing habitat reduces the rearing capacity of the habitat and may also lead to overwinter mortality (Chapman and McLeod, 1987).

Increased sediment delivery typically increases turbidity (Guy, 1970) by increasing suspended sediment levels. Although the relationship between suspended sediment varies by lithology and soil types, suspended sediment and turbidity are typically well correlated within a given watershed. Increased turbidity can impair the sight feeding of salmon at concentrations greater than 25 NTU (Lloyd et al., 1987). Gill damage can occur at high levels of suspended sediment.

Channel widening caused by increased sediment delivery is expected to aggravate seasonal temperature extremes (Theurer et al., 1984). Channel widening can also destabilize LWD because smaller pieces are unstable in wider channels (Bisson et al., 1987).

The maintenance of elevated sediment delivery inhibits the rate of recovery of degraded channel conditions degraded by increases in sediment delivery. Recovery of channel form from aggradation depends on a complex suite of factors including discharge, vegetation, soils, and sediment supply. However, if all other factors are kept equal, the rate of recovery from aggradation is partially a function of sediment delivery (Perkins, 1989). The recovery of degraded substrate conditions requires a sequence of flows with low sediment concentrations and that capable of flushing fine sediments via bed mobilization (Diplas, 1991).

3.2.2 Activities affecting sediment delivery: Sediment delivery is increased by activities that increase erosion and/or increase the efficiency of the transport of eroded sediment to streams. Erosion is increased by activities that remove vegetation and/or ground cover, disturb soils, increase overland flow frequency, or increase streamflows. Mining, road construction, grazing, developed recreation and grazing elevate erosion and sediment delivery through various mechanisms including soil compaction, removal of groundcover, removal of vegetation, increased runoff, increased peakflows, loss of bank stability, and concentration of surface runoff.

Logging and road construction elevate erosion considerably. It is estimated that logged areas in the Snake River Basin erode at about 2 to 3 times natural rates on a per unit basis (USFS, 1981; King, 1993). However, erosion increases from logged areas appear to decline to natural levels over time as revegetation occurs (USFS, 1981). Roads erode at 50 to more than 220 times natural levels during the construction phase (USFS, 1981; King, 1993). Although road erosion declines with time, roads continue to erode at levels several times greater than natural for the life of the road (USFS, 1981; Geppert et al., 1984). These increased levels of erosion are generally translated into increased sediment delivery. Based on a review of recent literature, Geppert et al. (1984) concluded that the construction, use, and maintenance of roads in Washington increases sediment delivery in 1st to 2nd order watersheds. MacDonald and Ritland (1989) also concluded that roads typically double suspended sediment yield even with state of the art construction and erosion control and that suspended sediment contributions from roads alone, even in the absence of mass failure, are typically in the range of 5 to 20 percent above background and remain at elevated levels for as long as roads are in use. Extensive clearcut logging and road construction in small experimental watersheds in the highly erodible Idaho batholith increased sediment yields by an average of 45 times (King, 1993). In the mountainous region of Montana, Anderson and Potts (1987) found that logging and road construction increased suspended sediment levels by seven times the background level in the first year after the activity and by two times the background level in the second year. On the eastside of the Washington Cascades, Fowler et al. (1987) documented that the construction of a road crossing increased turbidity by more than 50 times relative to an upstream site.

Logged and roaded areas also have much higher rates of mass failures than do undisturbed areas although the relationship varies among regions (Dunne and Leopold, 1978; Furniss et al., 1991). The mass erosion volumes originating in clearcuts ranges from about 1 to about 9 times that found in undisturbed areas in the coastal Northwest (Furniss et al., 1991). Roads have been found to increase mass erosion volume by about 30 to 350 times the amount occurring in undisturbed forested areas on a per unit area basis. In a study on the Idaho batholith, roads were found to have increased mass erosion by about 188 times the rate found in forested areas (Furniss et al., 1991). Dyrness (1967, as cited in Dunne and Leopold, 1978) found that the rate of mass failures from roads and logged areas was, respectively, about 500 and 10 times the rate occurring in undisturbed areas during a period of heavy rains and snowmelt in the Oregon Cascades. Dunne and Leopold (1978) concluded that road construction in most mountainous terrain would always promote increased landsliding, regardless of how much care was taken in planning and construction. Geppert et al. (1984) echoed this conclusion by stating, "The association of roads with debris avalanches is not specifically related to the construction phase or road use, but the fact that roads exist...Unlike failures within harvest units, the potential for debris avalanches from roads does not appear to decline with time except as the more susceptible areas fail." The frequency and volume of mass failures from roads depends strongly on climate, the paths of subsurface water, geology, and soils, as well as construction practices and maintenance (Furniss et al., 1991).

It appears that headward channel erosion in small, low order ephemeral channels caused by logging can be a significant source of elevated sediment delivery. Megahan and Bohn (1989) found that channel expansion and headward erosion by "stream sapping" in "zero order" channels was a significant source of sediment delivery caused by logging and road construction in their study in the Idaho batholith. On average, channel erosion from sapping failures was estimated to range from 1-25% of the total post-disturbance sediment yields from three small watersheds; Megahan and Bohn (1989) concluded that sapping was an important source of erosion.

Increases in peakflow can cause cross-sectional and headward channel expansion in ephemeral channels, even where riparian vegetation is left intact. In work on forested mountain streams in Arizona, Heede (1991) found that ephemeral stream channels increased their rate of headward erosion and expanded channel cross-sections in response to a 28% reduction in basal area and some road construction, in spite of buffers that extended approximately 150 feet on each side of the ephemeral streams. Heede (1991) concluded that the sediment delivery caused by the channel erosion was likely to have inconsequential downstream effects due to lagged delivery and channel storage; however, he failed to consider cumulative impacts from these sources. Based on Heede's

(1991) data, the increases in channel cross section caused by increased peakflows contributed approximately 550 yd^3 of sediment downstream over eight years excluding contributions from headward erosion just from the effect of logging upstream of ephemeral reaches that totaled less than 1.25 miles in length.

Grazing significantly increases surface erosion via soil compaction and the removal of groundcover vegetation (Lusby, 1970; USFS, 1980). In Colorado, Schulz and Leininger (1990) compared conditions within a thirty old exclosure and reaches that had reduced cattle grazing and found that the exclosure had twice the amount of litter cover, 20% the amount of bare ground, 8.5 times the willow canopy cover, and 5.5 times the shrub cover as the grazed areas. Grazing also increases channel erosion via trampling of banks and decreased bank stability caused by the loss of deep-rooted vegetation (Kauffman et al., 1983; Clary and Webster, 1989; Platts, 1991). Kauffman et al. (1983) found that bank erosion was more than three times higher in grazed reaches than within ungrazed reaches on Catherine Creek in northeastern Oregon.

Mining also elevates sediment delivery considerably, although it varies with the type of mining. Surface mining typically causes the greatest elevation in erosion and attendant sediment delivery. Nelson et al. (1991) concluded that erosion from surface mining and spoils is one of the greatest threats to salmonid habitats in the western U.S.

Given available information, it is not surprising that some of the most degraded conditions related to sediment occur in watersheds that have been logged, mined, roaded and heavily grazed, such as Johnson and Bear Valley Creeks (see Figures 5 and 16). As indicated in the figures, streams with high sediment loads have much higher fine sediment and cobble embeddedness and lower pool frequencies and estimated salmon STE than comparable streams with lower sediment loads

The effect of developed recreation on sediment delivery has not been extensively studied but it probably has effects similar to roads and road construction due to vegetation loss and soil disruption.

3.2.3 Evaluation: Available data plainly indicates that sediment delivery has a pronounced effect on salmon survival because it alters many vital aspects of salmon habitat. It is also clear that high levels of sediment delivery are a major constraint on salmon survival and production in spawning and rearing habitat in many streams in the Snake River Basin, including tributaries to the Clearwater, Tucannon, Salmon, and Grande Ronde Rivers.

As discussed in Section 1.1 Channel Substrate, limited data indicate that there is little prospect for the recovery of degraded substrate conditions unless estimated sediment delivery is reduced to less than 20% over natural. However, given environmental conditions in the Snake River Basin it is likely any elevation of sediment delivery over natural is likely to maintain elevated levels of surface fine sediment. The salmon habitats in the Snake River Basin are prone to sedimentation due to their location within watersheds, snowmelt-dominated hydrology, and highly erodible geology. As Chapman and McLeod (1987) noted in their review of the effects of fine sediment, any increases in fine sediment levels in rearing and spawning habitat should be avoided until accurate predictors of substrate response to sediment delivery and salmon response to increased fine sediment can be fully developed and tested; this concept means that resource managers should make "...every reasonable effort to reduce sediment recruitment from basin development" (emphasis added). We strongly concur with this assessment.

Sediment delivery is also important to control because its effects on salmon survival are only slowly reversible. Fine sediment readily and rapidly intrudes into gravels. Removal of fine sediment at depth requires both low sediment concentrations, and flows large enough to entrain bed materials (Diplas, 1991). Even once adequate efforts have been made to reduce sediment loading, the flushing of fine sediments will be slow. Therefore, it is critical to avoid levels of sediment delivery that can increase sedimentation.

As in Section 1.1 Channel Substrate, we recommend the following standards for watersheds where substrate standards are not met and sediment delivery is estimated to be more than 20% over natural: 1) Reduce sediment delivery through suspension of on-going activities and prohibition of the initiation of activities that can increase erosion over natural levels (See Table 1), and implement active restoration measures (e.g., road obliteration) as needed, until substrate conditions meet standards or a statistically significant ($p<0.05$) improving trend over at least 5 years is documented through monitoring and total sediment delivery from anthropogenic sources is estimated to be less than 20% over natural. 2) If substrate conditions do not meet standards after total sediment delivery is estimated to be less than 20% over natural and substrate conditions have exhibited a statistically significant, improving trend over at least five years, activities that can increase erosion should only be implemented/re-initiated when combined with active and passive restoration measures so that they result in net reductions in sediment delivery until substrate conditions meet standards.

We recommend the following for watersheds where substrate standards are not met but total sediment delivery from all anthropogenic sources is estimated to be less than 20% over natural: 1) Eliminate on-going activities and prohibit activities that can increase erosion over natural levels and implement active restoration measures, such as road obliteration, until substrate conditions meet standards or an statistically significant ($p<0.05$) improving trend over at least 5 years is documented through monitoring. 2) Once substrate conditions have exhibited a statistically significant trend over at least five years, activities that increase erosion should only be implemented/re-initiated when combined with active and passive restoration measures that result in net reductions in sediment delivery, until substrate conditions meet standards.

We recommend the following standards for watersheds where substrate standards are met but sediment delivery is estimated to be more than 20% over natural: Activities that increase erosion over natural levels should only be implemented or continued when combined with active and passive restoration measures that result in net reductions in sediment delivery, until sediment delivery is less than 20% or substrate conditions exhibit a statistically significant ($p<0.05$) improving trend over at least five years as documented by monitoring.

We recommend that factors of safety be incorporated into attempts to reach a net reduction in sediment delivery through active restoration combined with land disturbance. Anderson et al.

(1992) recommended that active restoration efforts should offset expected increases in sediment delivery by a factor of three. Due to uncertainties in the effectiveness of active restoration and the estimation of sediment delivery, we believe the threefold reduction in expected sediment delivery recommended by Anderson et al., (1992) is a prudent, minimum factor of safety regarding efforts to offset the sediment delivery from activities by active restoration. We also recommend that these active restoration should be completed and shown to be effective prior to initiating or continuing land disturbing activities, when net reductions in sediment delivery is the goal.

If substrate conditions show statistically significant deterioration ($p<0.40$) over any period longer than one year, activities that increase erosion should be suspended until substrate conditions return to their initial condition prior to deterioration; active restoration measures aimed at reducing sediment loads should also be undertaken in such cases. The level of low level of statistical significance ($p<0.40$) regarding habitat deterioration is recommended as part of a risk adverse approach and because of the environmental and economic costs associated with failing to address habitat degradation once it has occurred. (For more detail, see Section 1.6 Notes on Statistical Significance).

Until sediment delivery from grazing is incorporated into existing models, the recommended sediment delivery levels (i.e., reducing sediment delivery to less than 20% over natural) are based on the assumption that grazing will be suspended in degraded watersheds. In watersheds where grazing continues to occur, efforts should be made via field sampling and modeling to incorporate sediment delivery from grazing into estimates of total watershed sediment delivery.

The use of modeled estimates of sediment delivery as a management tool is fraught with the potential for error and abuse. However, available models have been widely used in the Snake Basin and, occasionally, validated. Despite the drawbacks inherent in a modeling-based approach, it appears to be the most viable tool for constraining land disturbance to levels that allow recovery and prevent degradation via sedimentation. Due to its widespread usage, we recommend that sediment delivery be initially estimated by models based on the USFS R1-R4 Sediment Yield Model (USFS, 1981; Potyondy et al., 1991) adapted so that sediment delivery from the following sources are included: 1) mass failure and surface erosion from logging; 2) mass failure and surface erosion from all existing roads, **regardless of age**; 3) channel and surface erosion from grazed lands; and 4) surface and mass erosion from mining. We also strongly recommend that efforts be taken to refine, validate, and calibrate the model locally.

However, we stress that in-channel habitat conditions are the bottom line. Although the USFS R1-R4 Sediment Yield Model appears to be the only readily and widely used tool available for constraining sediment loads to levels that may allow recovery in degraded systems and protect habitats from deleterious increases in sediment loads, the approach does have considerable weaknesses. The model (USFS, 1981; Potyondy et al., 1991) does not predict actual sediment delivery well, and was not meant to predict sediment delivery from specific events or years (King, 1993). Further, there is only fragmentary data indicating that degraded streams will recover if sediment loads are reduced to less than 20% over natural. While this appears to be the case based on the best available information, it is entirely possible that streams will not recover under our recommendations. Nonetheless, we see no other option for a prudent approach to reducing sediment delivery from activities except excluding all further land disturbance from watersheds and undertaking only active restoration measures aimed at abating sediment delivery. Given these critical uncertainties, biologically-based substrate standards must be the bottom line: where substrate conditions deteriorate or degraded conditions persist, efforts to reduce sediment delivery should continue until substrate sediment conditions meet the standards or exhibit a statistically significant, improving trend ($p<0.05$) over at least 5 years as documented through monitoring.

Where sediment delivery estimates are not available, we recommend that all on-going land disturbing activities should be suspended until sediment delivery is estimated.

3.3 LOGGING-RELATED DISTURBANCE: EQUIVALENT CLEARCUT AREA (ECA) APPROACHES

3.3.1 Effects on salmon: As previously discussed, a wealth of information indicates that timber harvest and roads reduce salmon survival and habitat productivity through increased sedimentation, alteration of basin hydrology, and the removal of riparian vegetation. Through a variety of mechanisms (See Figure 36), these changes generally lead to increased levels of fine sediment, loss of LWD and pools, channel widening, loss of structural channel diversity, summer water temperature elevation, and elevated peakflows. The magnitude of these management-induced changes generally increases as the magnitude of timber harvest and roads within a watershed increase. Available, but limited, data and information indicate that the amount of mobile fine sediment in pools within streams increases with increasing amounts of timber harvest and road construction (MacDonald and Ritland, 1989; Frissell, 1992; Lisle and Hilton, 1992; Hagberg, 1993). Frissell (1992) found that pool depths in specific channel types decreased with increased amounts of logging-related disturbance. Limited data also indicate that pool frequencies and LWD decrease with increasing amounts of timber harvest within watersheds (Reeves et al., 1993). Frissell (1992) found that LWD orientation in heavily logged systems was generally less stable and in positions less likely to form pools than in less disturbed systems, although he found no statistically significant difference in pool frequencies and LWD amounts related to logging levels. Lisle and Hilton (1992) and Hagberg (1993) found that the amount of pool volume filled by fine sediment was positively correlated with qualitative and quantitative estimates of sediment delivery. The estimated magnitude of sediment delivery generally increased with increased levels of logging and road levels, though harvest vintage and location, and basin geology were also factors (Lisle and Hilton, 1992). Reeves et al. (1993) found that salmonid diversity was decreased with increased timber harvest in southwestern Oregon, although the abundance of specific age classes and species of salmonids had limited statistical relationship to the amount of logging-related activities within the basins studied. Other work indicates that habitat quality decreases with increased levels of logging-related disturbance and sedimentation (Chen, 1992). As discussed, these changes in habitat condition caused by logging-related disturbance tend to reduce salmon survival and habitat productivity, especially when combined.

While available information indicates that the likelihood and magnitude of habitat alteration increases with increasing levels of logging-related disturbance, many other factors also exert strong controls on the amount and type of habitat damage caused by any given level of disturbance. A host of environmental factors influence the magnitude of the on-site impact for a given type of activity, how the impact is transmitted to the stream system, and the response of habitat conditions.

Generally, the proximity of the disturbance to streams is one of the strongest controls on the type, number, and magnitude of effects on habitat conditions. For instance, disturbance within riparian zones not only increases sediment delivery through a wider variety of mechanisms than does upland timber harvest, but also typically reduces stream shading and LWD amounts over time. The magnitude of these changes per unit area of disturbance generally increases with proximity to the stream channel, if all other factors are kept equal. The effects of land disturbance on habitat conditions is complicated because other factors besides disturbance magnitude also influence the magnitude and duration of the effects on downstream salmon habitat. Factors influencing habitat and channel response and recovery times include climate, channel morphology, stream network characteristics, riparian vegetation, bank and channel substrate, and basin hydrology.

Although erosion and sediment delivery tend to increase with increasing land disturbance within a given watershed, on- and off-site effects vary with harvest practice, climate, geology, soil characteristics, slope, vegetation, watershed location, elevation, stream proximity, and hydrologic effects that can also be affected by an array of factors. Although a complete and full review of these factors and their interrelationships is beyond the scope of this work, some discussion is warranted. For a given type and level of impact, surface erosion generally increases as the following increase: slope, soil erosivity, susceptibility of soils to compaction, precipitation, and amount of bare soil. In areas prone to mass failures from logging and roads, very small fractions of the watershed are estimated to produce the bulk of erosion related to forest management (Rice and Lewis, 1991). Channel gradient and tributary junction angle appear to influence the distance downstream that mass failures travel initially (Benda, 1988) before subsequent downstream transport by fluvial processes. The rate and efficiency of the sediment delivery to streams from surface erosion increases with increasing proximity to streams, increasing drainage density, increasing slope, and precipitation, other factors remaining equal. Sediment delivery from surface erosion decreases with increased amounts of vegetation, litter, ground cover, and land surface roughness, other factors remaining equal.

Logging-related disturbance in riparian zones disproportionally increases sediment delivery to downstream habitat, not only due to increased efficiency of sediment delivery of on-site erosion to streams, but also due to the number of mechanisms affected. The removal of riparian vegetation by logging and road construction increases channel erosion and channel widening by reducing bank stability and decreasing channel roughness via the removal of riparian zone vegetation and the loss of LWD over time. LWD loss also leads to the eventual downstream transport of stored sediment and the loss of available sediment storage sites. Removal of riparian vegetation also reduces the capacity of vegetation to detain sediment from upslope and upstream sediment sources, which may increase sediment delivery from these sources.

The type and age of disturbance also greatly affects erosion and sediment delivery. Roads generally cause far greater increases in erosion and sediment delivery than timber harvest, especially during the construction phase, other factors remaining equal. Erosion from roads initially decreases after construction, but then remains at levels far in excess of natural for the life of the road (USFS, 1981; Geppert et al., 1984). Logging methods, yarding methods, and site preparation, all influence the amount and duration of increases in erosion, because the various forestry activities differ in their effect on vegetative cover, root strength, local hydrology, and soils. Machine piling of slash and broadcast burning increase erosion beyond that caused by harvest and yarding (USFS, 1981). Erosion levels from logged areas decrease over time to nearly natural levels (USFS, 1981).

The downstream physical and biological effects of increased magnitude and duration of sediment delivery are expressed through interactions among several channel components. These include the amount of available sediment storage available in pools (Lisle and Hilton, 1992) and behind LWD obstructions (Megahan, 1982), channel gradient (Lisle and Hilton, 1992), channel cross-section (Lisle, 1982), particle size distribution of delivered and transported sediment (Dietrich et al., 1989; Lisle and Hilton, 1992), bed and bank materials (Richards, 1982; Carson and Griffiths, 1987; Howard, 1987; Rosgen, 1993), discharge regime (Everest et al., 1985; Perkins, 1989), and vegetation (Graf, 1979; Ikeda and Izumi, 1990). Low gradient, unconfined streams in alluvial valleys are the most sensitive to increases in sediment delivery causing increased levels of fine sediment (USFS, 1983; Frissell, 1992; Rosgen, 1993), pool in-filling (Frissell, 1992; Lisle and Hilton, 1992), and increased channel width (Schumm, 1969; Lisle, 1982; Richards, 1982; Jackson and Beschta, 1984; Howard, 1987; Carson and Griffiths, 1987; Lisle and Hilton, 1992). Frissell (1992) found that relationships between the amount of area logged and habitat changes were difficult to detect without a channel and watershed classification system, especially for changes caused by increased sediment delivery and peakflows.

3.3.2 Activities affecting the amount of logging-related disturbance in watersheds: Various methods and inventories have taken widely different approaches to determine the magnitude of disturbance levels within watersheds. Some methods treat logged and roaded areas separately (McCammon, 1993), while others use weighted coefficients to include the area of roads in the "equivalent clearcut area", or to include logged areas as part of a "equivalent roaded area" (Coburn, 1989). Some methods treat road area and harvest area as equivalent (L. Bach, Umatilla National Forest Hydrologist, pers. comm., 1993; P. Boehne, Wallowa-Whitman National Forest Fish. Bio., pers. comm., 1993; R. Jones, Clearwater National Forest Hydrologist, pers. comm., 1993). Most approaches include areas logged less than some arbitrary age in estimates of the amount of watershed area disturbed by logging. McCammon (1993) suggests using only harvest units less than 30 years old; the Umatilla National Forest considers only harvest units less than 10 years old in its disturbance estimates (L. Bach, Umatilla National Forest Hydrologist, pers. comm., 1993). The La Grande Ranger District of the Wallowa-Whitman National Forest includes all identifiable units beneath a basal area threshold (P. Boehne, Wallowa-Whitman National Forest Fish. Bio., pers. comm., 1993). The disturbance index method proposed by Klock (1985) includes only logging units less than 10 years old. These various approaches are generally premised on assumptions about the amount and duration of change caused by the various types of disturbance, e.g. that logged units older than 10-30 years have fully recovered all ecological functions significant to downstream conditions.

There are also various approaches for amalgamating logging disturbance by amount of overstory removal. The Umatilla National Forest includes only clearcut and shelterwood harvest in its estimates of harvest area and ignores selectively harvested areas (L. Bach, Umatilla National Forest Hydrologist, pers. comm., 1993) even though these areas are often extensively entered and compacted. On the La Grande Ranger District, remaining basal areas after harvest are used to convert selection harvest units to equivalent clearcut units (P. Boehne, Wallowa-Whitman National Forest Fish. Bio., pers. comm., 1993). A similar approach is used on the Clearwater National Forest (R. Jones, Clearwater National Forest Hydrologist, pers. comm., 1993). McCammon (1993) provides no explicit direction on how to treat various types of harvest in estimating the amount of a watershed with units less than 30 years old. All of these approaches assume that partial cuts cause less on- and off-site change than clearcuts. The previously discussed approaches do not attempt to account for differences in yarding methods or site preparation. Klock (1985) proposed a set of arbitrary, but internally consistent, coefficients for weighting logging related disturbance based on activity type, slope, and age of activity, to calculate a "risk index" based on the amount and type of logging related disturbance. Although the methods vary as to how the affected area disturbed by logging and roads is calculated, we will refer to the generic conceptual approach as the "equivalent clearcut area" (ECA) approach.

3.3.3 Evaluation: Although it is highly desirable to limit the amount of logging-related disturbance to levels that avoid or prevent the degradation of downstream habitat, we do not recommend using watershed level ECA thresholds for logging-related disturbance based solely on the amount of watershed area affected. The ECA approach does not have much utility in protecting salmon habitat because it masks habitat damage in several key ways and is based on faulty assumptions. The ECA approach implicitly assumes that there is limited aquatic damage caused by the cumulative effects of logging and roads beneath some threshold area affected within a watershed. While this assumption is more tenable for disturbance outside of riparian zones, it is clearly not tenable for activities within riparian zones. Available information and data clearly indicate that relatively small amounts of vegetation disturbance in riparian zones damage aquatic habitat considerably.

A second, and related, problem with the ECA approach is that does not adequately address spatially varying factors that strongly control the magnitude of impact per unit area of disturbance, such as the damage caused by activities in riparian zone, or the amount of road in unstable terrain. All disturbance areas are treated equivalently even though they clearly do not lead to equivalent amounts of disturbance. For instance, ECA methods treat 2000 acres of a clearcut on ridges the same as 2000 acres of clearcut riparian area. Disturbance within riparian zones clearly is a critical element for diagnosing existing and potential cumulative effects on habitat conditions. Because ECA approaches are "blind" to riparian disturbance, the approach has limited utility as a tool for habitat protection or diagnosing cumulative effects.

Use of watershed-level ECA estimates completely obscures the amount of riparian disturbance, and, hence, habitat damage. This critical flaw, alone, renders the ECA approach unusable because logging and roads in riparian zones disproportionally damage fish habitat with respect to both the magnitude and number of conditions degraded. For example, consider a watershed where riparian zones constitute 6% of the watershed area. In such a watershed, ECA magnitudes of 4% may cause devastating impacts to habitat even if only half of the 4% watershed ECA is located in riparian zones, because this would translate to having approximately 33% of the entire riparian area in a logged or roaded condition. This level of riparian disturbance is sure to result in severe sedimentation, elevated water temperatures, and the loss of LWD and pools.

The failure of the ECA to adequately capture riparian disturbance is of still greater concern because field reviews and available data indicate that riparian zones have been entered and roaded in watersheds that have been logged (See Figure 33). In many systems, riparian areas have been disproportionally disturbed. Under these conditions, watershed scale estimates of logging related disturbance are misleading. Data from the Blue Mountain Province watersheds in Washington and on the Clearwater National Forest indicate that the fraction of riparian area harvested and/or roaded exceeds the fraction of the watershed area logged or roaded (Figure 33). Data from these watersheds indicate that the fraction of the riparian area in a clearcut, shelterwood, or roaded condition is, on average, about 1.48 times the fraction of the total watershed in equivalent condition. Notably, the Umatilla National Forest data do not include units that have been selectively harvested. The inclusion of selectively harvested areas may exacerbate the bias indicated because riparian areas have often been selectively harvested over the last ten years. The pattern of riparian disturbance is not surprising. Review of field conditions consistently indicates that main haul logging roads were primarily built along streams. Trees were consistently logged along these roads due to ease of access. Thus, ECA approaches inherently obscure the level of disturbance in areas where the magnitude of habitat damage per unit area of disturbance is greatest and where, in some cases, the amount of disturbance has been concentrated.

ECA approaches also do not adequately incorporate ecological recovery times and the duration of impacts when the disturbance is in riparian zones. As mentioned, existing ECA approaches do not include logging disturbances older than 30 years old. In contrast, the time to ecological recovery after logging requires 25 to 200 years, in riparian zones (Gregory and Ashkenas, 1990). Therefore, ECA approaches ignore areas causing on-going cumulative effects in riparian zones.

ECA approaches also fail to account for all activities at the watershed scale that contribute to the cumulative degradation of salmon habitat. ECA approaches address only roads and timber harvest and ignore mining and grazing. Both mining and grazing can severely damage riparian zones, water quality, and fish habitat, in the complete absence of timber harvest. The methods also fail to address synergies between grazing and logging-related disturbance. Field reviews consistently show that roads and timber harvest in riparian zones have greatly increased riparian zone grazing and concomitant habitat damage by facilitating livestock access. Loss of bank-stabilizing vegetation by grazing can exacerbate the effects of logging and roads. The failure of the ECA approaches to address grazing and mining renders it useless as a tool to avoid degradation or provide some index of the risk of cumulative effects in watersheds in the Snake River Basin. Grazing is a widespread source of habitat degradation. Grazing damages almost every aspect of salmon habitat, because grazing pressure is often greatest in riparian zones (Platts, 1991). Surface mining is among the most damaging activities per unit area (Nelson et al., 1991). Therefore, the ECA approach clearly fails to capture all land use activities that can cause cumulative effects and its use masks many sources of habitat damage.

Available data illustrate the failure of the ECA approach to capture sources of cumulative habitat degradation. Elk Creek on the Boise National Forest, a heavily grazed stream in wilderness, has about 40% fine sediment and is estimated to have sediment delivery at about 60% over natural, while in adjacent, ungrazed watersheds in wilderness have about 22% fine sediment (Boise National Forest, 1993). Bear Valley Creek on the Boise National Forest has severely degraded substrate conditions that have clearly reduced salmon survival profoundly (See Figures 1, 2, and 5). Over the past 50 years, the amount of fine sediment has almost doubled and about 57% of the large pools have been lost in Bear Valley Creek (Boise National Forest, 1993). These conditions are largely due to accelerated sediment delivery that is currently estimated to be at about 115% over natural (Boise National Forest, 1993). Grazing, historic mining, and logging-related disturbance are estimated to contribute sediment delivery at 60%, 50%, and 5% over natural, respectively (Boise National Forest, 1993). Streams in watersheds in the Idaho batholith that have been grazed, mined and logged are much more degraded with respect to pool frequency and substrate conditions than systems that have only been logged (See Figures 5, 16, 37).

Data from the grazed subwatersheds in the Blue Mountain Province also indicate that poor habitat conditions exist at a wide range of ECA levels (See Figures 10, 13-15, 19, 20, 22, 28, 29, 34, 38, and 39). Notably, site-specific evaluations concluded that grazing has contributed to the existing degradation in the Upper Grande Ronde River (Beschta et al., 1991; Anderson et al., 1992; Umatilla National Forest, 1993; McIntosh et al., 1994). Hence, ECA methods are probably not well-suited to conditions in the Blue Mountain Province. Almost all habitat conditions in the Upper Grande Ronde River watershed were not significantly correlated to ECA level in grazed subwatersheds in the Blue Mountain Province (See Figures 14, 15, 19, 20, 22, 28, 29, 34, 38, and 39), again indicating that ECA approaches appear to have limited utility in areas with considerable riparian disturbance and a history of grazing. In the Upper Grande River watershed, the only habitat variable that was significantly correlated to ECA level was the amount of channel substrate dominated by sand (R^2=0.14; $p<0.10$).

Estimates of sediment delivery correlate very poorly with ECA levels in watersheds in the Idaho batholith (R^2=0.21; $p>0.10$ (See Figure 40)). This may partially explain why some habitat conditions are not well correlated with ECA levels. However, bank stability was significantly correlated with ECA levels (R^2=0.55, $p<0.05$) and ECA levels did explain more of the variability in pool frequency (R^2=.34, $p>0.10$) than did estimated sediment delivery in watersheds in the Idaho batholith (See Table 3).

It is apparent that significantly degraded habitat conditions that are undesirable for the recovery of the listed salmon species result from disturbance levels that are well below thresholds assumed to confer a "moderate to low risk" of cumulative effects on aquatic habitats. For instance, it has been assumed that ECA levels of less than 15% of the watershed confer a low risk of cumulative effects to streams (McCammon, 1993). However, degraded habitat conditions occur at ECA levels below 15% (See Figures 7, 8, 21, 34, and 37). In the Blue Mountain Province of Oregon, streams draining grazed watersheds with ECA levels of 10% and 12%, had maximum daily water temperatures of almost 80°F during the summer of 1992 (Umatilla National Forest, 1993); such water temperatures can cause direct mortality in salmon (Armour, 1991). While ECA levels explained some of the variability in bank stability and pool frequency in the Idaho batholith, it did not explain much of the variability in substrate conditions and sediment delivery (See Table 3) which are major habitat problems in the Idaho batholith. Almost all habitat conditions in the Blue Mountain Province in Oregon were uncorrelated with ECA levels (See Figures 14, 15, 19, 20, 22, 28, 29, 34, 38, and 39) making it difficult to set thresholds that protect aquatic resources. The lack of correlation with habitat conditions may be due to the ECA approach's inability to incorporate spatial considerations, impact duration, all disturbance impacts, and habitat sensitivity.

It is probably not possible to establish ECA thresholds beneath which habitat damage does not occur or the risks of cumulative effects are not high due to the inherent weaknesses in the ECA approach. Available information, data and field reviews consistently indicate that a considerable amount of watershed level disturbance has occurred within riparian zones. Available information indicates that a "no impact" threshold of disturbance in riparian zones probably does not exist. This is probably also true of watershed-level disturbance as indicated by available data. Sensitive systems, such as the South Fork of the Salmon River, underwent catastrophic and long-lasting degradation at an ECA of less than 15% (D. Burns, Payette National Forest Fish. Bio., 1993). Streams in grazed watersheds can have very degraded conditions at an ECA level of zero. Most importantly, many habitat conditions at a wide range of ECA values in logged and grazed watersheds in all regions were poor. Therefore, the ECA approach clearly has limited utility in developing protective thresholds of watershed development. This is a critical flaw because the utility of an ECA approach in protecting salmon habitat is premised on the ability to identify thresholds that reliably prevent cumulative damage to salmon habitat over time.

Due to inherent flaws, ECA approaches cannot be used to estimate levels of logging-related disturbance that are compatible with allowing the recovery of degraded habitat conditions. Current ECA approaches address only the "risk" of cumulative impacts. They do not address the need for habitat recovery in degraded systems. Available trend data indicate that degraded stream substrate conditions do not begin to recover even at relatively low levels of ECA where there is on-going disturbance, even with significant in-channel enhancement efforts of questionable effectiveness. For instance, Deadman Creek on the Clearwater National Forest exhibited a statistically significant (R^2=0.57; $p<0.10$) increase in fine sediment by depth from 1985 to 1992 (Clearwater National Forest, 1993) with watershed ECA levels ranging from 4 to 9% of the watershed area during the trend period (Clearwater National Forest, 1994 (See Figure 8)). Over the same time period, there was no statistically significant improvement (R^2=0.004; $p>0.10$) in fine sediment by depth in Pete King Creek on the Clearwater National Forest with ECA levels ranging from 13 to 16% of the watershed area (Clearwater National Forest, 1994 (See Figure 7).

We conclude that current ECA approaches are not amenable for use in developing thresholds of logging-related disturbance that will effectively avoid and prevent habitat damage and allow recovery in degraded systems. A more promising approach to avoiding cumulative degradation of habitat is to fully protect riparian zones and floodplains, limit upland disturbance on the basis of the estimated effect on the driving variables that affect habitat conditions, such as sediment delivery, in concert with the coarse screening approach recommended in this report, e.g. assessing the consistency of land management activities with habitat improvement based on habitat status relative to the recommended habitat standards and compliance with land use standards.

3.4 ROADS

3.4.1 Effects on salmon: Available information consistently indicates that roads are one of the greatest sources of habitat degradation in managed watersheds, especially when they are within riparian zones (Geppert et al., 1984; Furniss et al., 1991). Roads significantly elevate on-site erosion and sediment delivery for the life of the road (USFS, 1981; Geppert et al., 1984; USFS et al., 1993). Studies consistently indicate that roads increase the frequency of mass failures in mountainous terrain (Dunne and Leopold, 1978; Megahan et al., 1978; Geppert et al., 1984; Furniss et al., 1991). Mass failure volumes from roads are orders of magnitude greater than from undisturbed areas on a per unit area basis (Dunne and Leopold, 1978; Megahan et al., 1978; Geppert et al., 1984; Furniss et al., 1991). Road crossings cause extreme increases in sediment delivery (Fowler et al., 1987). Roads also disrupt subsurface flows (Megahan, 1972). Roads increase peakflows (King and Tennyson 1984; King, 1989; USFS et al., 1993). Roads within riparian zones reduce shading and disrupt LWD sources for the life of the road (USFS et al., 1993). These effects of roads degrade habitat by increasing fine sediment levels, reducing pool volumes, increasing channel width and exacerbating seasonal temperature extremes.

3.4.2 Activities affecting roads: A review of the activities affecting roads is outside of the scope of this report.

3.4.3 Evaluation: Existing data amply indicate that roads and road construction have been a major cause of habitat degradation in the Snake River Basin (See sections on channel substrate (1.1), pools and LWD (1.2.1), riparian reserves (3.1), sediment delivery (3.3), and logging-related disturbance (3.4)). Roads clearly disrupt watershed and riparian function (USFS et al., 1993). The on- and off-site effects are only very slowly reversible.

We recommend that no additional roads should be constructed in Snake River Basin watersheds with salmon habitats until 90% of the salmon habitats in managed watersheds have met substrate standards or have shown a statistically significant ($p<0.05$) improving trend over at least five years as documented by monitoring. We recommend that existing road mileage in all watersheds should be reduced, over time, especially within riparian reserves.

We also recommend that roads that will not be re-located or obliterated should be improved to reduce sediment delivery and improve drainage, except where the hazards of improvement pose short-term threats to salmon that may not be outweighed by the long-term benefits of road improvement. In such situations, roads should be closed/obliterated/relocated.

3.5 GRAZING

3.5.1 Effects on salmon: Data clearly indicate that livestock grazing has numerous adverse effects on fish habitat including channel widening, shade reduction, elevated sedimentation, and exacerbation of seasonal water temperature extremes.

Grazed land erodes at rates considerably higher than ungrazed areas (Dunne and Leopold, 1978; USFS, 1980). Sediment yields from grazed watersheds are significantly higher than those from ungrazed watersheds (Lusby, 1970). Much of the increased sediment delivery from grazed watersheds is due to the loss of vegetation which accelerates streambank erosion. Streambank erosion and subsequent sediment delivery is higher in grazed areas than ungrazed areas (Kauffman et al., 1983; Platts, 1991). The loss of riparian vegetation often leads to gullying and stream incisement (Graf, 1979; Schumm et al., 1984; Harvey and Watson, 1986) which causes significant increases in erosion and downstream sediment delivery (Schumm et al., 1984; MacDonald and Ritland, 1989). High levels of channel erosion due to bank instability and vegetation loss have been consistently noted in grazed areas in Oregon and Idaho in evaluations of channel and habitat conditions (J. Rhodes, unpublished field notes, 1989; Beschta et al., 1991; Beschta et al., 1993; Rhodes et al., 1993). Grazing strategies that have been judged to be somewhat compatible with the recovery of deep-rooted vegetation may be the least compatible with protection of bank stability from trampling. Early season grazing has been opined to be compatible with allowing vegetative regrowth of shrubs in riparian zones (Elmore, 1992). However, in Montana, Marlow and Poganick (1985) found that cattle damage to streambanks was strongly correlated to soil moisture content (R^2=0.85). Thus, early spring/summer grazing strategies are probably not consistent with protecting streambanks from damage because of high soil moisture content during these periods.

Riparian grazing may also increase sediment delivery by reducing riparian vegetation. Riparian vegetation is one of the most effective means of reducing erosion and sedimentation caused by land disturbance (Megahan, 1984a; Heede et al., 1988). Riparian vegetation also traps sediment along channels which aids in building banks and restoring channel morphology (Platts, 1991). The interception of sediment delivery by vegetation is critical in most managed drainages because rates of erosion and sediment delivery have been increased by the combined effects of a variety of ground-disturbing activities that accelerate erosion and sediment delivery.

Riparian grazing widens channels through multiple mechanisms because it causes increased bank erosion, loss of bank vegetation, loss of bank stability, and increased sediment delivery in combination. Each of these effects individually renders streams more prone widening. Stream sections in grazed areas are often much wider than in ungrazed areas (Platts, 1991). The width-to-depth ratio of streams often decreases over time once livestock access has been eliminated and bank stabilizing vegetation is re-established (Platts, 1991).

Grazing can lead to pool loss by elevating sediment delivery (Lisle, 1982; Lyons and Beschta, 1983; Jackson and Beschta, 1984; Alexander and Hansen, 1986; MacDonald and Ritland, 1989; Lisle and Hilton, 1992), reducing wood inputs from hardwoods, and by decreasing bank stability. Over the past 50 years, pool loss has been significant in grazed watersheds such as Bear

Valley Creek (Boise National Forest, 1993) and the Grande Ronde River (McIntosh, 1992; McIntosh et al., 1994 (See Figures 10 and 11)). Once channel morphology has been altered by sedimentation, the recovery rate is partially dependent on the amount of on-going sediment delivered to the stream system (Platts and Nelson, 1985; Platts et al., 1989; Perkins, 1989; Bohn and Megahan, 1991). Even instream structural enhancement to build pools is unsuccessful, if upstream sediment delivery from grazing is not reduced (Platts and Nelson, 1985). Cattle can cause pool loss directly by shearing off undercut banks; pools associated with undercut banks are important habitat components for rearing juvenile and migrating adult salmon, especially in meadow systems.

Grazing can also increase fine sediment levels by increasing sediment delivery. In Johnson and Bear Valley Creeks, accelerated erosion from unstable banks and channels appears to be one of the primary contributors to the high fine sediment levels that have resulted in extremely low levels of salmon survival (Boise National Forest, 1993; NMFS, 1993). Bank stability on Bear Valley Creek appeared to be positively related to forage utilization (Boise National Forest, 1993). It is estimated that grazing has elevated sediment delivery by about 60% over natural in Bear Valley Creek. Although fine sediment levels appear to be inversely related with forage utilization levels, it is doubtful that reduced forage utilization, alone, can result in reductions in fine sediment levels. Fine sediment levels cannot improve unless total sediment delivery from all sources is reduced to less than the transport capacity of the stream. Available models (USFS, 1983) indicate that fine sediment levels in Bear Valley Creek are likely to increase under existing sediment loads, including those from grazing. Given available information, sediment delivery may have to be reduced to at least 20% over natural if recovery of substrate conditions is to occur in Bear Valley Creek, as discussed in the sections on channel substrate (1.1) and sediment delivery (3.2)

Habitat conditions in grazed and ungrazed tributaries to the Middle Fork of the Salmon River provide an example of the effects of grazing on salmon survival. Currently, salmon STE is estimated to be about 0.5% in Johnson Creek (NMFS, 1993), and about 3% in Bear Valley Creek (Boise National Forest, 1993; Scully and Petrosky, 1991), while STE averages about 29% in ungrazed tributaries draining wilderness areas (Scully and Petrosky, 1991) although the survival estimates may be somewhat biased by rapid emigration of juvenile salmon from Bear Valley Creek into less degraded streams.

Grazing reduces stream shading by reducing the size and frequency of riparian vegetation through trampling and forage use or by shifting the composition of plant communities. The reduction of stream surface shading by the removal of riparian vegetation is typically the single greatest cause of water temperature increases in managed watersheds (USFS, 1980; Theurer et al., 1984; Beschta et al., 1987). Mountain streams without riparian vegetation are often too cold in the winter for salmonids and are also very susceptible to the development of icing (Platts, 1984; Platts, 1991). Stream icing can cause direct mortality of overwintering fish and eggs in redds (Platts, 1984; Boise National Forest, 1990). In some cases, streams can be pirated away from established channels by livestock trails, dramatically reducing stream shading (J. Rhodes, unpublished field notes, 1990). Although comprehensive data are lacking, field reviews and data indicate that stream shading has been reduced due to the loss of vegetation in many grazed reaches in the Snake River Basin (J. Rhodes, unpublished field notes, 1989; Beschta et al., 1991; Beschta et al., 1993). The loss of riparian vegetation caused by grazing may also reduce water temperatures by increasing air temperatures over streams. Channel widening can also exacerbate seasonal temperature extremes.

Grazing can also aggravate seasonal water temperature extremes by reducing baseflows via channel incision and/or soil compaction. Field reviews indicate that channel incision leading to water table lowering is common in grazed areas in parts of Idaho (Beschta et al., 1993) and Oregon (Beschta et al., 1991; J. Rhodes, unpublished field notes, 1989; 1991; 1992; 1993; Rhodes et al., 1993). Groundwater inputs during winter and summer moderate seasonal water temperature extremes, because groundwater tends to be warmer than streamwater during winter and colder than the streamwater during the summer. Loss of baseflows also renders streams more susceptible to warming by reducing stream discharge. Restoration of riparian soils and vegetation through improved range management is one of the most viable management tools available for increasing summer baseflows (Ponce and Lindquist, 1990). Winegar (1977; 1978, as cited in Reeves et al., 1991) found that fencing of a stream in eastern Oregon increased summer baseflows and ceased the consistent freezing of the stream during winter.

The effects of grazing on salmon habitat are persistent. Long-term trend data on the South Fork of the Salmon River in Idaho indicate that even with a moratorium on land disturbance in conjunction with active sediment reduction measures, the recovery of stream substrate conditions has not been complete after 25 years (Platts et al., 1989; Bohn and Megahan; 1991). Notably, improvement in fine sediment conditions may be slower in grazed systems such as Bear Valley and Johnson Creeks than in the South Fork Salmon River because fine sediment levels and sediment delivery currently are far higher than they were in the South Fork Salmon River when recovery began to occur. On the west side of the Cascades, the recovery of water temperature and stream shading on ungrazed, forested streams takes at least 25 years after disturbance has ceased (Gregory and Ashkenas, 1990; Beschta et al., 1987). Vegetative recovery will be probably slower in the Snake River basin due to generally slower rates of vegetative growth. In some heavily degraded systems, vegetative recovery of deep-rooted deciduous species that provide stream shading appears to be delayed by the persistent effects of soil compaction and competition with grasses (E. Claire, ODFW Fish. Bio., pers. comm., 1993). In these heavily degraded streams, the regrowth of deciduous woody vegetation may not begin until several years after grazing elimination (E. Claire, ODFW Fish. Bio., pers. comm., 1993).

Ungrazed streams or reaches typically have greater numbers of salmonids than grazed reaches or streams (Stuber, 1985; Platts, 1991; Rich et al., 1992). Rich et al. (1992) found during six years of monitoring that the average numbers per unit area of juvenile steelhead and chinook salmon were about 20 and 10 times higher, respectively, in ungrazed control streams than in streams degraded by grazing (Bear Valley Creek, as discussed above). Grazing elimination in riparian areas has been demonstrated to improve habitat or habitat attributes in most studied cases (Kauffman et al., 1983; Stuber, 1985; Schulz and Leininger, 1990; Platts, 1991; E. Claire, ODFW Fish. Bio., pers. comm., 1993). In Central Oregon, the suspension of grazing increased stream shading from alder and willow by 75% in 10 years (Claire and Storch, 1983 as cited in Platts, 1991). Clifton (1989) estimated that the removal of cattle from a highly degraded riparian zone in Central Oregon reduced stream channel width by more than 30% over 16 years. Little recovery occurs where exclosures are small and upstream grazing remains heavy (Platts and Nelson, 1985). Studies indicate that grazing elimination increases the numbers of salmonids using a reach, although some aspects of the studies render these results questionable (Platts, 1991).

Riparian grazing also can have direct effects on salmon, because livestock can trample redds when fording streams during the incubation period. Trampling of redds can cause partial or complete mortality of the incubating eggs.

3.5.2 Activities affecting grazing: Although there is a whole host of environmental and economic factors affecting grazing levels and distribution, a review of these factors is beyond the scope of this report. However, it is worth noting that other land management activities do influence livestock grazing pressure in riparian areas. For example, field reviews consistently indicate that the removal of riparian trees and road construction increase grazing pressure in riparian zones.

3.5.3 Evaluation: Although comprehensive data are lacking, existing data indicate livestock grazing has contributed to the degradation of habitat in the Snake River Basin. Complete elimination of riparian area grazing is the grazing strategy most compatible with re-vegetation and salmon habitat recovery. Existing information indicates that there is a low probability that any grazing management system will allow recovery to begin in damaged riparian systems without some initial period of rest (Clary and Webster, 1989; Platts, 1991). Most available, widely-used grazing practices are incompatible with the protection and restoration of aquatic ecosystems. While it has been suggested that there are grazing strategies that are compatible with the recovery of degraded riparian systems (Elmore, 1992), there appear to be limited data to corroborate this claim; data do not appear to exist that indicate that any riparian grazing strategy can result in the same <u>rate</u> of recovery of riparian vegetation and channel conditions as can be achieved with elimination of riparian grazing (W. Elmore, USBLM Riparian Specialist, pers. comm., 1990). Field reviews and available information amply indicate that continued grazing retards the recovery of vegetation and channel morphology in degraded riparian systems (Platts, 1991; Green, 1991; Beschta et al., 1991; J. Kauffman, Ore. State Univ. Prof. of Rangeland Resources, pers. comm., 1992; J. Rhodes, unpublished field notes, 1993; Rhodes et al., 1993; Boise National Forest, 1993 (See Figure 32)). Grazing also appears to reverse vegetative recovery. Schulz and Leininger (1990) found that density of woody species was significantly less in reaches where grazing had been re-introduced than within exclosures; they also found that canopy cover by willow was about 8.5 times greater within the exclosures than in grazed reaches. Evaluations of fish habitat conditions affected by grazing, at scales ranging from the reach to regional, have repeatedly recommended the temporary or permanent elimination of riparian grazing in degraded riparian areas in order to initiate and/or accelerate the recovery of riparian vegetation, channel conditions, and fish habitat conditions (Clary and Webster, 1989; Beschta et al., 1991; Anderson et al., 1993; Henjum et al., 1994). Skovlin (1984, as cited in Clary and Webster, 1989) recommended at least five years of rest for degraded areas prior to re-introducing grazing under proper management.

As with other types of land disturbance that cause increased erosion, we recommend that grazing be suspended in watersheds that do not meet substrate standards until the standards are met, or a statistically significant ($p<0.05$) improving trend over the course of 5 years is documented through monitoring and total sediment delivery is estimated to be less than 20% over natural. Livestock grazing in watersheds where water temperature standards are not met in salmon habitat should be suspended within the riparian reserves until water temperature standards are met, or a statistically significant ($p<0.05$) improving trend over at least 5 years is documented through monitoring. In watersheds where bank stability standards are not met, we recommend that grazing be suspended within half of a tree height from the edge of the floodplains or from the edge of streams where floodplains do not exist, until the bank stability standard is met, or monitoring documents that a statistically significant ($p<0.05$) improving trend has occurred over at least five years. We recommend that these same approaches should be taken if habitat conditions set as standards exhibit a deteriorating trend between years ($p<0.40$). In many areas, riparian area grazing is difficult to control; in these areas it will be necessary to completely remove livestock from watersheds in order to preclude grazing within floodplains and reserves until recovery occurs or standards are met.

We also recommend that grazing be eliminated from environments where it is clearly incompatible with the protection of aquatic resources. While it is not currently possible to completely identify all such environments, it is relatively clear that grazing in perennially saturated meadows with fine-grained, non-cohesive soils and without woody bank vegetation almost always leads to stream damage. Therefore, we recommend that such environments not be subjected to grazing.

Where grazing continues or is re-initiated after vegetation, soils, and habitat conditions have recovered, grazing use should be light. In non-degraded riparian areas along reaches and in watersheds where habitat standards are met, grazing should be suspended until allotments plans are revised to be as compatible with protection of vegetation, channels, and soils and monitoring is in place. We recommend mandatory monitoring of habitat variables set as standards in affected riparian areas that are grazed and in downstream habitat. Exclosures should be used on all active allotments as monitoring references, as others have recommended (Beschta et al., 1991; Anderson et al., 1992; Beschta et al., 1993). Where degradation or deviation from trends within exclosures is documented, grazing should be suspended until recovery has resumed.

The suspension of grazing in degraded reaches and watersheds has the greatest promise of any restoration measure for attaining rapid improvement in habitat conditions and salmon survival within the next ten years. Grazing is widespread in the Snake River Basin and so is attendant degradation. Grazing is estimated to be one of the single largest sources of management-induced sediment delivery in the Snake River Basin and the single largest nonpoint source in Idaho. The suspension of grazing poses no risk to ecosystems and has been consistently demonstrated to be effective in triggering measurable improvement in habitat conditions and fish populations.

Although forage utilization rates have some utility in limiting damage from grazing (Clary and Webster, 1989), we do not recommend sole dependence on lowered forage utilization rates as a measure of increased habitat protection. Henjum et al. (1994) concluded that forage utilization levels have little established ecological relevance to aquatic resources. Field reviews indicate that forage utilization standards are an ineffective approach to restoration and protection in degraded reaches, wet meadows, seeps, and travel corridors because habitat damage is caused by trampling and chiseling of banks and vegetation by livestock rather than forage utilization. A better approach is to eliminate grazing in these areas. The control of forage utilization, alone, probably does not adequately address bank trampling, soil compaction, sedimentation and restoration of riparian plant assemblages and status. In a three year study in Montana, Marlow and Poganick (1985) found that bank damage from trampling was uncorrelated with forage utilization levels ($R^2 = 0.06$), but was strongly correlated with soil moisture content in soils ($R^2=0.85$).

In order to prevent the trampling of redds by livestock, we recommend that livestock be excluded from riparian areas along all spawning reaches prior to spawning and for the duration of the incubation period.

3.6 ROADLESS RESERVES

3.6.1 Effects on salmon: Through a variety of mechanisms, the cumulative effects of activities that disturb vegetation and soils can lead to increased levels of fine sediment, loss of LWD and pool volumes, channel widening, loss of structural channel diversity, summer water temperature elevation, and elevated peak flows. As discussed, these changes in habitat condition caused by logging-related disturbance tend to reduce salmon survival and habitat productivity, especially when combined (See sections on channel substrate (1.1), pools and LWD (1.2.1), riparian reserves (3.1), sediment delivery (3.3), and logging-related disturbance (3.4))

Many managed watersheds have degraded stream conditions leading to reduced salmon survival. Degradation has occurred via sedimentation caused by mining, logging, and road construction in the tributaries of the Middle Fork of the Salmon River (Boise National Forest, 1993) and in the Grande Ronde River (McIntosh, 1992; Anderson et al., 1992). Salmon survival has been severely reduced in Bear Valley Creek due to sedimentation caused by the combined impacts of grazing, mining, roads and logging (Scully and Petrosky, 1991; Boise National Forest 1993 (See Figure 1 and 2)). Middle Fork Salmon River tributaries adjacent to Bear Valley Creek that are within wilderness/roadless areas, have much less fine sediment and higher salmon survival and densities than Bear Valley Creek. Pool losses have been significant over a 50-year period in all managed basins that have been resurveyed in the Snake River Basin, except the Tucannon, which has a considerable portion of the watershed in a wilderness/roadless condition. Over the same 50-year period, pool loss was insignificant in streams with watersheds in a wilderness/roadless condition (McIntosh, 1992 (See Figures 10-14)). The documented loss in pools is probably due to the combined effects of increased sediment loading and the loss of LWD over time. Although comprehensive data are lacking, mining, roads, and logging within riparian areas has also reduced LWD throughout much of the Snake River Basin. The loss of riparian shading caused by grazing, logging, and roads in riparian areas have contributed to the adversely high water temperatures existing in the Tucannon, Grande Ronde, and Clearwater tributaries.

The catastrophic sedimentation of the South Fork of the Salmon contributed to the precipitous declines of salmon populations in the river that may have once been the most productive chinook salmon habitat in the entire Snake Basin (Platts et al., 1989). Although sediment conditions have improved, recovery has not been complete and salmon survival in the South Fork Salmon River remains depressed due to fine sediment conditions (Platts et al., 1989). The catastrophic degradation of the South Fork Salmon River was caused primarily by mass failures from logging roads and logged areas; it is estimated that less than 15% of the watershed had been disturbed by logging and roads at the time of the mass failure events (D. Burns, Payette National Forest Fish Bio., pers. comm., 1993). Although passage mortality at downstream dams has contributed to the decline of spawning populations of salmon in the South Fork Salmon River, it is clear that habitat degradation and attendant reduction in survival in natal habitat have had a major role in reducing the salmon populations of the South Fork Salmon River (Platts et al., 1989; Petrosky and Schaller, *in press*).

In the Snake River Basin, the geographic scale of habitat degradation has undoubtedly contributed to the declines in salmon populations at the basin scale. Much of the Snake River Basin habitat outside of roadless and wilderness areas has been degraded as indicated by evaluations of habitat conditions at a variety of scales and locations. Platts and Chapman (1992) concluded that about 61% of the spawning and rearing habitat in the Salmon River Basin was in poor condition, based on available information and field knowledge. Data from the Clearwater National Forest (1991) indicate that about 71% of managed watersheds had adversely high levels of sediment. The Upper Grande Ronde River has also been affected by mining, splash dams, significant road construction, grazing, logging, and a fire/flood event. These combined activities have contributed to the poor habitat conditions existing in the Grande Ronde, including high levels of fine sediment and cobble embeddedness, low levels of LWD and pools, high summer water temperatures, frequent winter icing events, low levels of stream shading, and low bank stability (Anderson et al., 1992).

Salmon survival in spawning and rearing habitat integrates stream conditions, which themselves integrate landscape level processes such as nutrient cycling, watershed hydrology, and sediment flux. Terrestrial disturbance alters natural processes in ways that are not fully predictable, nor completely understood (Grayson et al., 1993). The magnitude and duration of the effect of disturbance on habitat conditions will differ widely depending a dizzying array of ecosystem characteristics at levels ranging from the reach (gradient, substrate, etc) (Rosgen, 1993) to the watershed (climate, geology, vegetation, etc).

The consequences of entry into undisturbed systems are probably lowly reversible. Although accelerated surface erosion may only persist for 6-10 years, hydrologic alteration may persist for more than 20 years (Harr and Coffin, 1990). Accelerated erosion from roads, as well as other effects, persist for as long as the roads exist, and then some. Even after obliteration, roads continue to erode at levels far in excess of natural for several years (Potyondy et al., 1991). Functions provided by large downed wood, such as terrestrial sediment storage, require more than 100 years after trees have been removed for recruitment to be re-established. The prospects for recovery of channel morphology and sediment cycling are extremely poor for steep headwater streams in non-cohesive soils that have been degraded, even if the cause of degradation is arrested (Rosgen, 1993).

3.6.2 Activities affecting roadless reserves: Mining, logging, and road construction reduce the amount of area reserved in an undisturbed condition or in a state of recovery. Grazing also alters vegetation composition and soil conditions.

3.6.3 Evaluation: Given available data and known linkages among land use effects and habitat conditions, it can be reasonably concluded that the best water quality and habitat conditions needed by salmon exist in roadless/wilderness areas or where continuing disturbance of roadless areas has not occurred. For instance, although the South Fork Salmon River has not fully recovered, surface fine sediment levels (about 10-15% (Idaho Dept. of Health and Welfare, 1991)) in the South Fork Salmon River are lower than other managed watersheds in the Idaho batholith (See and compare Figures 5 and 9). In fact, levels of surface fine sediment in Johnson Creek and Bear Valley streams exceed the amount of surface fine sediment existing in the South Fork Salmon River at its worst during the 1960s. It is estimated that average surface fine sediment in the South Fork Salmon River peaked at about 47% (Platts et al., 1989). Johnson Creek currently has about 63% surface fine sediment (NMFS, 1993), and Bear Valley Creek has about 56% surface fine sediment (Boise National Forest, 1993).

A moratorium on logging-related disturbance has been effective in decreasing fine sediment levels in the South Fork Salmon River, but over the last several years, fine sediment conditions have ceased to improve. The cessation of improvement in the South Fork Salmon River appears to be due to continuing levels of elevated sediment delivery from the remaining road system and other sources (Platts et al., 1989). Nonetheless, it appears that the substrate conditions in the South Fork Salmon River are much better than in the bulk of watersheds in the Idaho batholith that have been roaded, logged, mined and/or grazed.

Once degradation has occurred and roadless areas have been mined, roaded, and logged, there is limited potential for recovery. Available trend data from the Clearwater National Forest indicate that there has been little or no recovery in streams with high levels of fine sediment as more roadless areas are logged and entered, even with continuing and substantial in-channel efforts to remove fine sediment from the streams. Data from Pete King Creek on the Clearwater National Forest (1992) indicate that percent fines by depth do not show long-term statistically significant improvement over the trend period (slope coefficient = 0.15% fines by depth/yr; R^2=.004; $p >> 0.10$), although the data appear to indicate short-term cycles of improvement and deterioration (See Figure 7). Notably, significant upstream efforts to remove fine sediment via sediment traps and excavation have been in place in Pete King Creek since 1987. Substrate data from Deadman Creek on the Clearwater National Forest show a deteriorating trend in fine sediment conditions when averaged over the entire trend period (slope coefficient= 2.54 % fine sediment by depth/yr; R^2=0.57, $p<0.10$ (See Figure 8)). Data from Lolo Creek on the Clearwater National Forest also show a statistically significant deteriorating trend during the monitoring period in fine sediment conditions in cleaned gravels placed to provide winter habitat (See Figure 4). Lolo Creek has been heavily degraded by elevated sediment delivery from logging, roads, and mining (Espinosa and Lee, 1991). Available trend data in fines by depth at Lolo Creek also show no improvement in degraded conditions during the monitoring period (See Appendix B). Substrate conditions have also deteriorated over the past 50 years in Bear Valley Creek; recent trend data do not indicate recovery (Boise National Forest, 1993).

Despite existing data, many have speculated that roadless areas can be entered without degrading habitat conditions via careful planning, avoidance of high risk areas such as riparian areas to the extent considered feasible, and implementation of "Best Management Practices" (BMPs). However, BMP effectiveness remains a matter of speculation. Most studies of the effects of BMPs have been too short in duration to capture lagged effects or provide an indication of long-term effects. Little is known about the cumulative effectiveness of BMPs in the face of significant landscape alteration. While many assessments of BMPs have focused on estimating the short-term reduction in accelerated pollutant loading, most studies have not examined whether aquatic habitat is fully protected over the long term (USEPA, 1993). There has been extremely limited assessment of the cumulative effectiveness of BMPs. Appendix B contains a thorough historical perspective on how the reliance on BMPs and unwarranted planning assumptions led to the continued degradation of salmon habitat on the Clearwater National Forest.

Available information indicates that adequate protection of riparian areas may do much to protect riparian functions such as LWD loading (McDade et al., 1990; Robison and Beschta, 1990) and shading (Beschta et al., 1987; USFS et al., 1993). However, protection of bank stability and shading within riparian zones may be ephemeral due to increased incidence of blowdown along disturbed areas (Franklin and Forman, 1987). Further, riparian protection may not eliminate adverse habitat alteration caused by the effect of logging and road construction on sediment delivery (Megahan and Bohn, 1989) and runoff (King, 1989; Megahan and Bohn, 1989; Heede, 1991). In degraded areas, such effects either exacerbate poor habitat conditions or prolong the time needed for recovery. Weak stocks with low survival may not recover from either additional degradation or maintenance of poor habitat conditions.

Scientific evaluations (Anderson et al., 1993; USFS, 1993b; USFS et al., 1993; Henjum et al., 1994) have consistently noted that roadless, unlogged tracts form the cornerstones of habitat recovery efforts. Existing roadless and wilderness areas provide the only high-quality habitats and islands of natural functioning systems left in the entire Snake River Basin. The extent of these areas is limited. It has not been shown under ecologically applicable experimental conditions that it is possible to enter roadless systems without compromising their natural function and/or without degrading habitat conditions, over time. Logging of roadless areas puts efforts to protect non-degraded and degraded habitats at risk (USFS et al., 1993).

Environmental conditions in roadless/wilderness areas also make their protection important. Many roadless/wilderness areas are in steep, erosive terrain with relatively high levels of precipitation and with snowmelt-dominated hydrology (The Wilderness Society, 1993; Henjum et al., 1994), rendering remaining roadless areas more prone to management-induced increases in peakflow (MacDonald and Ritland, 1989) and fine sediment in salmon habitat (Everest et al., 1987).

There is also considerable uncertainty about the effectiveness of restoration efforts in highly damaged watersheds, in terms of both rate and magnitude. Notably, USFS et al. (1993) concluded that tools were not available to characterize the expected effectiveness of protection/restoration measures in allowing habitat to improve to levels considered to assure viability of salmonid populations at risk; USFS et al. (1993) estimated probabilities of species viability were assessed via "expert" opinion.

It is not currently possible to make accurate forecasts of the cumulative effects of anthropogenic disturbances on salmon habitat. Fully mechanistic approaches to ascertaining the effect of disturbance fail due to data insufficiency and incomplete information and knowledge about processes and stress/response mechanisms. Some have argued that accurate prediction of changes in runoff caused by disturbance is currently impossible, regardless of analytical effort, because "physically-based" hydrologic models have no real physical basis due to the lack of understanding of actual watershed function aggravated by the lack of methods for collecting data at a meaningful scale (Beven, 1989; Grayson et al., 1993). However, in aggregate, available information indicates that increased landscape disturbance and the development of previously undisturbed areas increase the risk that salmon habitat will be degraded via alteration in watershed scale hydrology and material transfers. Given what is known about the effects of land disturbance on habitat conditions together with the existing inability to accurately predict cumulative effects it is not prudent to introduce disturbance into undisturbed areas. The defects of recent land management history described in Appendix B can be reduced with incorporation of better scientific information. But these defects cannot be eliminated as long as land management activities with irreversible negative effects continue to be initiated. Undisturbed areas within watersheds or the Snake River Basin as a whole not only provide habitat refugia for salmon, they also support continuance of natural linkages between terrestrial and aquatic systems. It is unlikely that habitat degradation can be arrested and reversed without protecting and restoring the natural functions of watersheds (USFS et al., 1993). Undisturbed areas (provided they are minimally influenced by alterations to upstream areas or neighboring watersheds) provide zones where the natural terrestrial processes can operate, and through linkages with the stream system produce habitat conditions typical of those in which the salmon evolved. Therefore, loss of undisturbed areas combined with slow or limited progress in reversing degradation across the majority of streams both within watersheds and regionally will further reduce aggregate numbers of salmon produced in the Snake River Basin. It also diminishes the potential for net habitat improvement over time that can contribute to the recovery of the listed salmon populations.

Mining, logging, and road construction in roadless areas have the potential to cause additional habitat loss at the Snake River Basin scale by degrading water quality in areas where it is currently high, further degrading downstream reaches, and forestalling habitat recovery. The prospect of reduction in habitat quality in currently high quality areas combined with the potential for maintenance of poor habitat conditions in degraded areas will seriously jeopardize the prospects for the recovery of salmon populations at the river basin scale, especially because the effects of entry into roadless areas are not immediately reversible and constitute a commitment of resources that is irretrievable. Continued diminishment of areas that maintain natural functions increases the risk of failing to improve habitat conditions at scales ranging from the reach to the region.

Given the existing extent and magnitude of habitat degradation, uncertainties about the prospects of habitat improvement in heavily damaged watersheds, and the irreversibility of effects, it is prudent to require that most of the degraded habitat be improved prior to taking <u>any</u> risks with the scarce areas with high quality habitat. As noted by Henjum et al., (1994) roadless tracts greater than 1000 acres are ecologically important because of existing forest fragmentation and widespread degradation. Therefore, we recommend that roadless tracts greater than 1000 acres should not logged or roaded, at least until substantial habitat recovery has been documented in most habitat conditions in managed watersheds in the Snake River Basin. In order to provide some assurance that some habitats remain as refugia for geographically isolated spawning populations, it is prudent to protect existing roadless areas at least until habitats in 90% of the managed watersheds in the Snake River Basin either meet all habitat standards or have exhibited statistically significant ($p<0.05$) improving trends over at least five years as documented through monitoring. In the event that degraded habitats do not improve and/or salmon are extirpated from some watersheds, refugia afforded by roadless tracts can provide some level of protection to populations that could ultimately recolonize barren salmon habitat. Smaller roadless areas may also be ecologically important. Therefore, we also recommend that smaller roadless areas should not be logged or roaded unless it can be demonstrated via peer-reviewed scientific study that it will have no negative effect on watershed conditions and will not foreclose management options for the recovery of salmon habitat.

3.7 GEOGRAPHIC CRITERIA FOR RE-EVALUATING LAND USE STANDARDS

3.7.1 Evaluation: We recommend using "adaptive management" concepts in the application of the screening process. That is, management approaches should be adjusted as more knowledge is gained and conditions change. Given this approach, the standards for land use set in the screening process should be subject to revision as conditions change. Specifically, less restrictive land use standards may be considered after habitat conditions have improved considerably. However, premature relaxation or over-relaxation of protection measures carries with it the risk of reversing or preempting habitat improvement. Protection of the habitat quality of refugia can maintain an important source of salmon colonists should habitat not improve in some watersheds due to severe and lowly reversible degradation, ineffectiveness of protection and restoration measures, or a combination of both. Careful thought should be given to developing criteria to define a reasonable point at which higher risk approaches may be considered. We recommend that the land use standards should only be revised once efforts to improve degraded habitats have been successful over a wide range of geography and conditions and should be further strengthened if habitat conditions do not improve.

We recommend that the land use standards should not be revised to allow activities with higher risk or that foreclose management options until habitat conditions in at least 90% of the managed watersheds in the Snake River Basin either meet all biologically-based habitat standards, or show a statistically significant ($p<0.05$) improving trend over at least a five-year period as documented by monitoring.

There are several reasons behind the recommendation. First, it is prudent to ensure that habitats and survival have improved significantly prior to foreclosing management options via entering roadless areas, altering vegetation within riparian reserves, or undertaking higher risk activities, especially given the biologically perilous condition of the listed salmon species, the existing widespread degradation of habitats, and uncertainty regarding improvement in habitat and survival. Once 90% of the managed watersheds have been documented to have improved, additional risks are at least slightly more tenable. Second, it is prudent to ensure that evaluation of habitat trend in each watershed is data driven and that recovery is geographically robust when watershed scale trends are aggregated at the basin scale. Our recommendation is aimed at requiring that there is a strong indication that restoration and protection efforts have been effective across a fairly wide range of geography. Improvement in the majority of habitats in managed watersheds provides some assurance that the continued application of combined measures can ultimately affect improvement in habitats that have not improved, if additional recovery time is allowed. Third, higher risk approaches should not be considered until improved habitats are well-distributed throughout the basin in order to provide refugia and protection for groups of potential salmon colonists. Restoration and long-term viability of salmon species will necessitate re-emergence of the full set of life history characteristics typical of the species that can only be elicited by presence of high quality, diverse, geographically widespread, spatially linked habitat units extending from headwaters to large rivers. Population stability will require presence of high quality habitat for all life stages and multiple habitat options needed by mobile species (i.e., increased habitat redundancy as opposed to the singular, isolated habitat units currently available). Population viability will be minimized as long as high quality habitat is restricted to high elevation, high gradient streams that may be naturally marginal habitats. Fourth, there should be some indication that there is greater system resiliency in habitat conditions throughout the basin. Our recommendation provides some assurance that individual habitats are more resilient due to watershed recovery and that the system of habitats at the basin scale has more resilience through system redundancy, in case some habitats are degraded by the re-initiation of higher risk activities or natural catastrophes.

Re-evaluation of land use standards should be made extremely carefully. The history of land use, habitat damage, and the response of salmon populations amply indicate that any gains in habitat conditions, salmon survival, and salmon numbers are rapidly reversible while losses are not. Under no circumstances should it be considered acceptable again to initiate activities that singly and cumulatively damage habitat. We do not recommend relaxation of land use standards, rather, we only suggest a rational criterion for success, after which, riskier approaches could be considered if coupled with careful monitoring to detect degradation. It should be remembered that our recommendations are not the lowest risk approach. Although we have reversed much of the traditional burden of proof regarding land use and habitat effects, we have not reversed all of it and instead have relied on a fairly mechanistic approach based on existing information. It should be kept in mind that there is no good <u>scientific</u> reason for <u>not</u> reversing <u>all</u> of the traditional burden of proof in protecting and restoring salmon habitat.

4.0 SCREENING AT THE WATERSHED SCALE OR FOR INDIVIDUAL ACTIVITIES

The coarse screening process can be used at the watershed scale to determine the consistency of cumulative activities with the goals of protecting and improving salmon habitat and survival. The process should be applied at watershed scales that represent logical units of salmon production and encompass both rearing and spawning habitat. Generally, experience indicates that the watersheds of streams that range from 3rd to 6th order at the watershed outlet can serve as logical production/management units. In some situations it may be desirable to apply the coarse screening process at smaller and larger watershed scales due to the salmon's use of the habitats or environmental conditions within a watershed. Thus, the actual scale of application will have to be dictated by field conditions. In many cases, the choice of watershed scale for analysis of cumulative effects on salmon habitat has already been made (e.g., the Upper Grande Ronde on the Wallowa-Whitman National Forest). In many cases, these previously selected scales should suffice for application of coarse screening process. We recommend that the core set of land management standards be applied as screening elements at all scales, on streams of all orders, to determine the consistency of activities with the goal of protecting and improving salmon habitat and survival.

The coarse screening process can also be used to evaluate the consistency of a single activity with the goals of protecting and improving salmon habitat and survival. However, watershed scale data for habitat conditions set as standards that can be affected by the activity are still required to apply the process for a single activity.

When screening at the watershed scale, data must be available and integrated on a watershed basis for all habitat variables set as standards. If data do not exist, all on-going activities that can potentially affect the parameters should be suspended or deferred until habitat data are collected and summarized. The potential effects of activities can be found in Table 1.

Data are required for land use parameters set as standards. If data do not exist, all activities that disturb vegetation and soils should be suspended or deferred until data are collected on land use status. Data requirements related to land use standards include the status and condition of roadless reserves, riparian reserves, roads, sediment delivery, and how current land use activities relate to the land management standards, e.g., whether on-going activities are within the riparian reserves.

The spatial distribution of habitat and watershed conditions is a key information need. A geographic information system data base is recommended for presenting data at the watershed scale, although it is not required as part of the screening process.

Where data indicate that habitat standards are not met, activities and that can prevent or forestall habitat recovery should be considered to be inconsistent with improvement and protection of salmon habitat and should be suspended or deferred. Table 1 provides an overview of activities and their potential effects on habitat conditions. Complete passive restoration at the watershed scale should proceed until trend data indicate habitat recovery. Where likely to be effective, active restoration measures should be undertaken to improve habitat conditions.

Where data indicate that an activity does not comply with standards for land management, it should be considered inconsistent with habitat protection and improvement and be modified to be consistent with land management standards, curtailed, or deferred. Active and passive restoration should be taken to move watershed conditions towards compliance with standards, unless trend data indicate that habitat standards are met or that habitat conditions are improving.

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APPENDIX A

FIGURES AND TABLES CITED IN THE TEXT

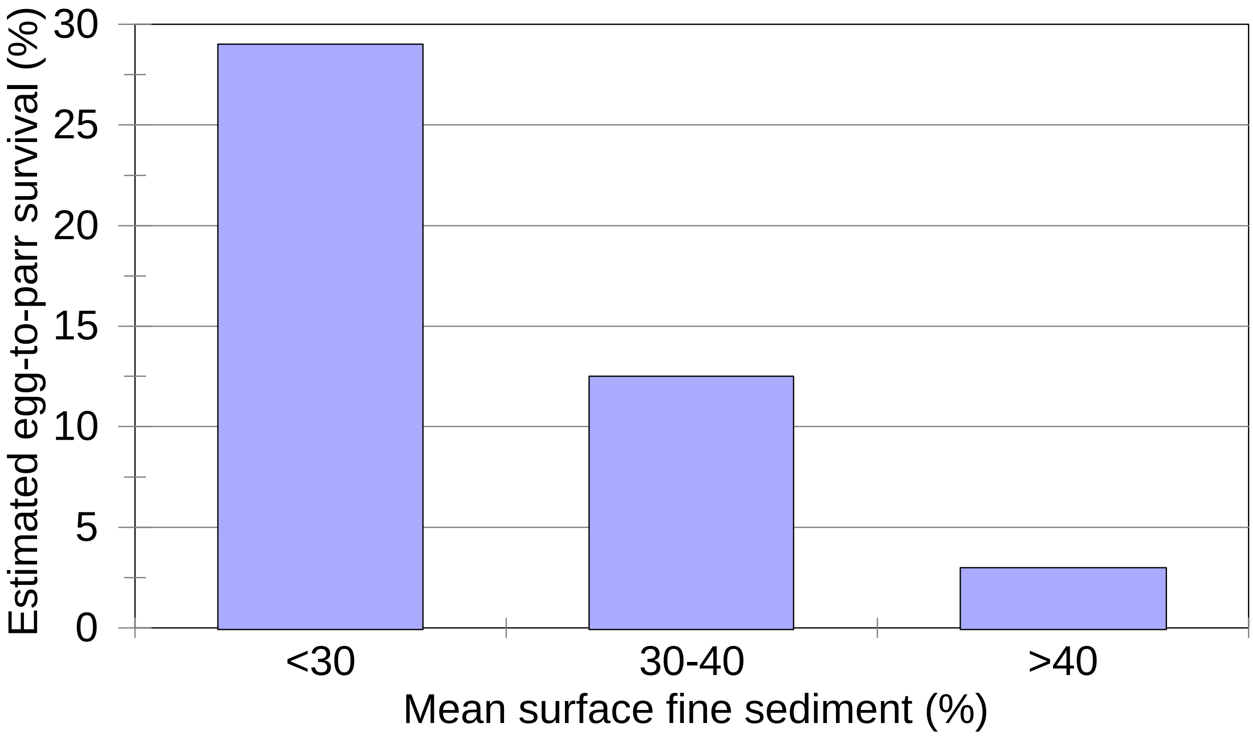

Figure 1. Average estimated spring chinook survival from egg to parr in Middle Fork Salmon River tributaries by category of percent surface fine sediment. Data averaged from Scully and Petrosky (1991). Survival rates for the <30% fine sediment category determined from one year of sampling on Salmon River and Marsh Creek. Survival rates for the 30-40% fine sediment category are averages from one year of sampling on Sulphur Creek and two years on Herd Creek. Survival rates for the >40% fine sediment category are averages from three years of sampling on Elk Creek and four years of sampling on Bear Valley Creek (Scully and Petrosky, 1991). It is possible that survival estimates for Bear Valley/Elk Creeks may be biased due to rapid emigration of juvenile salmon from areas with high surface fine sediment to areas with low surface fine sediment (C. Petrosky, IDFG Staff Bio., pers. comm., 1994).

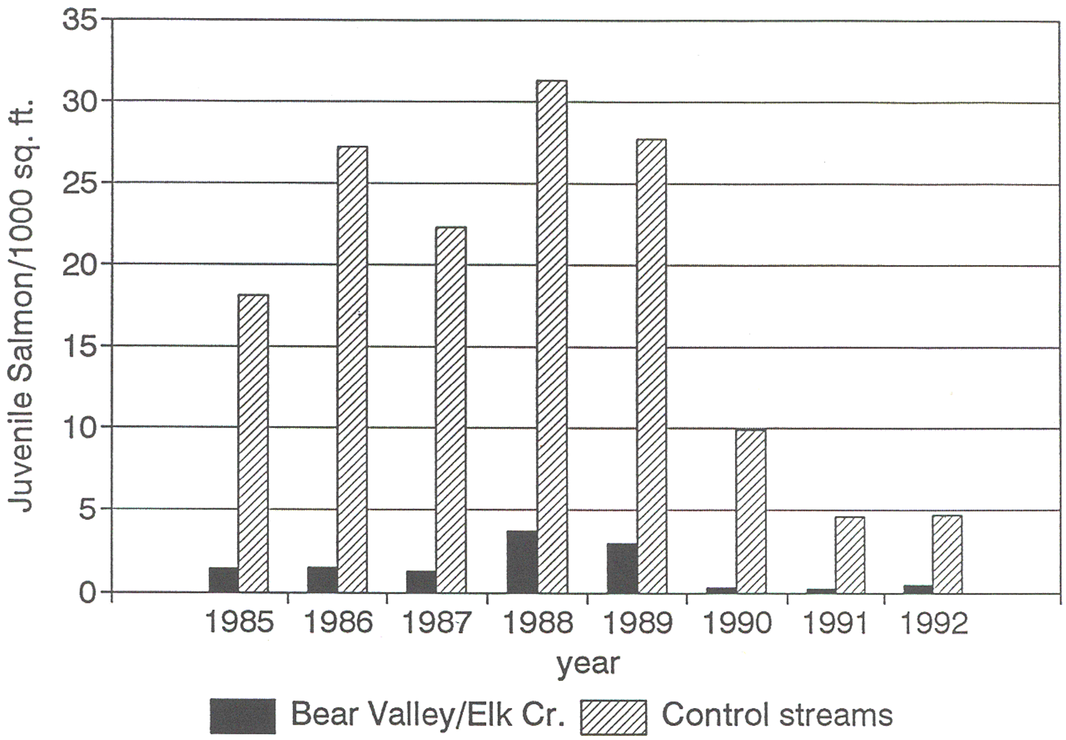

Figure 2. Average density of juvenile spring chinook by year in the heavily sedimented Bear Valley and Elk Creeks and in less sedimented control reaches. All reaches are in streams that are tributary to the Middle Fork of the Salmon River (MFSR). Control reaches are Chamberlain Creek, a tributary to the MFSR, and in reaches within the MFSR mainstem (C. Petrosky, IDFG Staff Bio., pers comm., 1994) All control reaches are ungrazed and drain wilderness. Control reaches average 20% surface fine sediment. Bear Valley/Elk Creek are both heavily grazed and sampled reaches averaged 46% surface fine sediment (Rich et al., 1992). The Bear Valley Creek watershed has also been logged, roaded, and mined. All stream reaches are managed solely for production of wild fish. The consistent differences between salmon densities is probably due to the density independent effect of surface fine sediment on salmon survival that occurs even at low seedings above eight mainstem dams. It is possible that the differences in densities in Bear Valley/Elk Creeks and the undegraded control streams may also reflect the rapid emigration of juvenile salmon from areas with high surface fine sediment to areas with low surface fine sediment (C. Petrosky, IDFG Staff Bio., pers. comm., 1994). (After Rich and Petrosky, 1994).

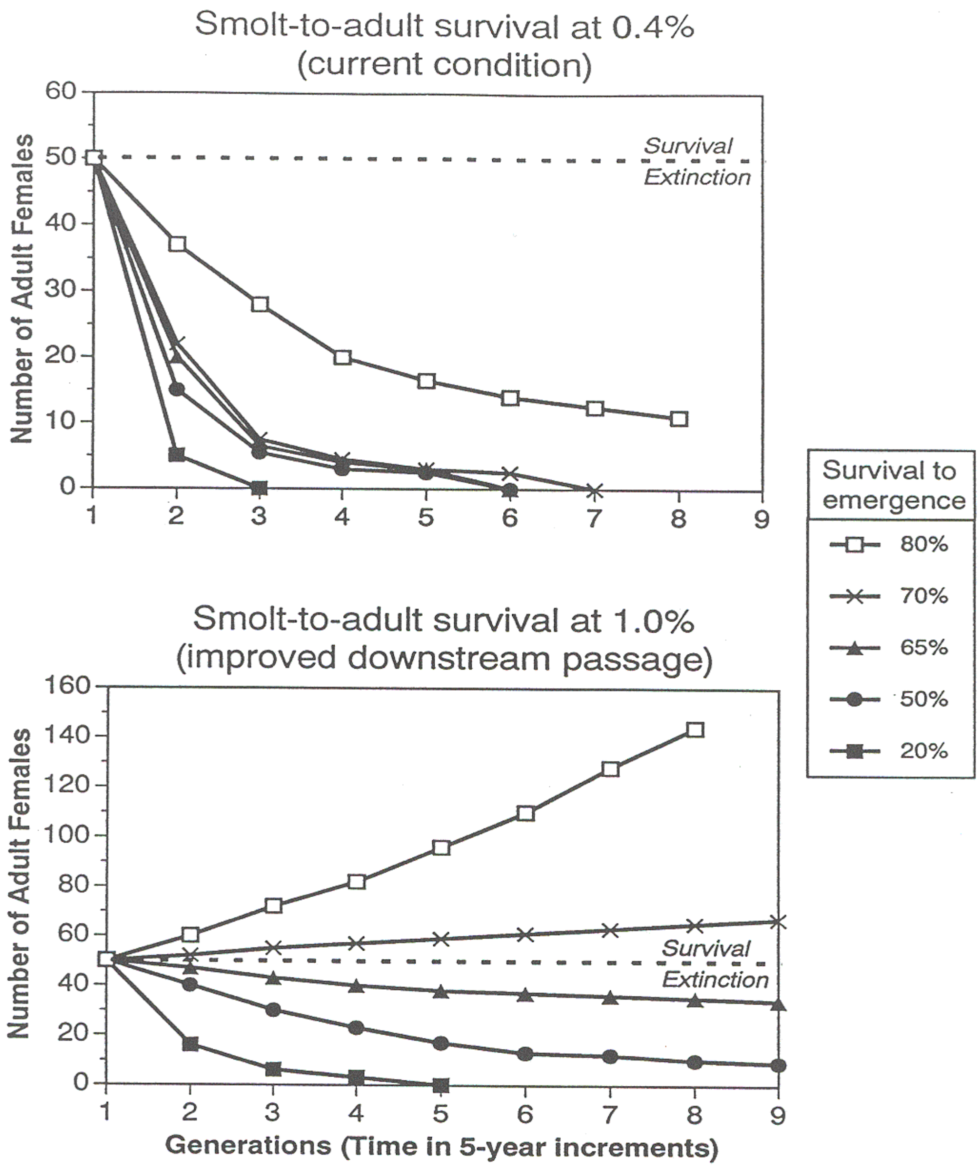

Figure 3. The effect of reduced survival-to-emergence (STE) on salmon population decline and time-to-extirpation in spring and summer chinook. Modeled output from USFS (1993a) based on assumed fecundities of females. The density-independent effect of reduced STE exacerbates the effect of downstream dam mortality on population decline. Some degraded streams in the Snake River Basin have extremely low STE levels (<5%) due to high levels of sedimentation (See Figure 1). If sediment conditions are allowed to persist in concert with high levels of mortality at downstream hydroelectric facilities, continued declines and rapid extirpation of spawning populations can be expected in many highly degraded drainages, even if mainstem passage is improved. (After USFS, 1993a).

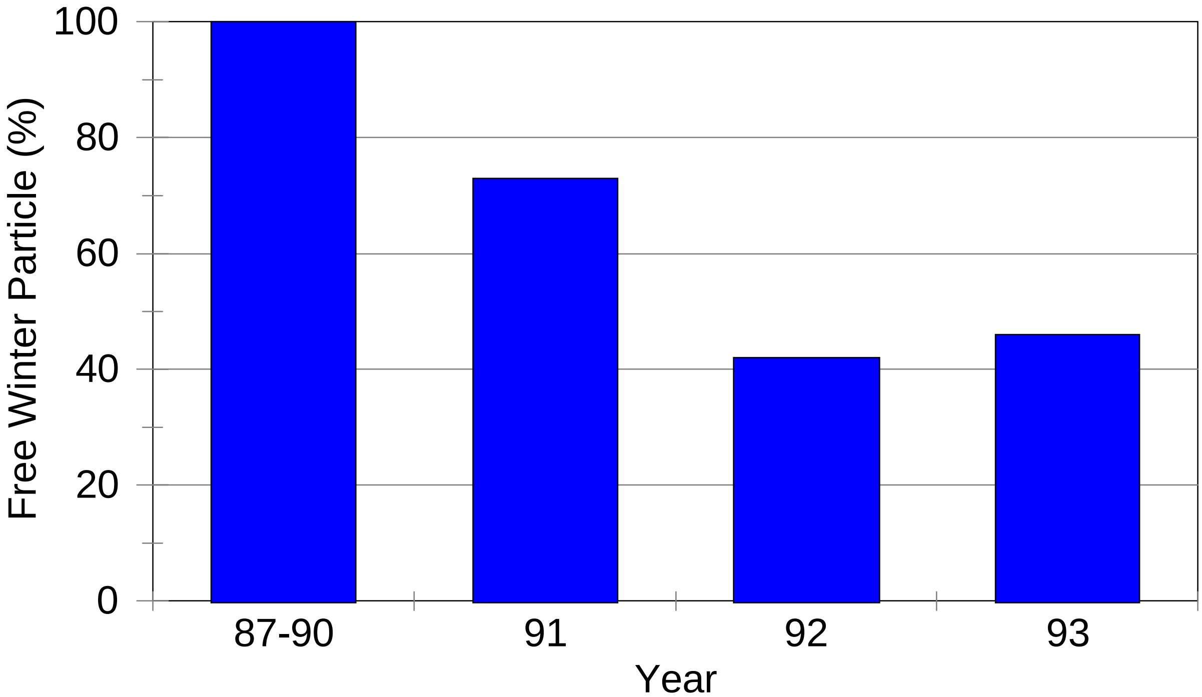

Figure 4. Trend in "free winter particle" over time in Lolo Creek on the Clearwater National Forest (CNF). Free winter particle is a measure of interstitial rearing space in winter habitat. Decreases in free winter particle also indicate in an increasing trend in cobble embeddedness (CE). Clean cobbles and boulders were added to the sample reach and the percent free winter particle (amount not embedded) was tracked over time. Data indicates that sedimentation continues to occur in winter rearing habitat under existing levels of sediment delivery. In 1991, sediment delivery in this reach of Lolo Creek was estimated to be about 37% over natural (CNF, 1991a). However, sediment delivery is probably underestimated because CNF sediment models ignore sediment contributions from roads more than 6 years old. Equivalent Clearcut Area was estimated to be about 26% of the watershed area (CNF, unpublished WATBAL runs).

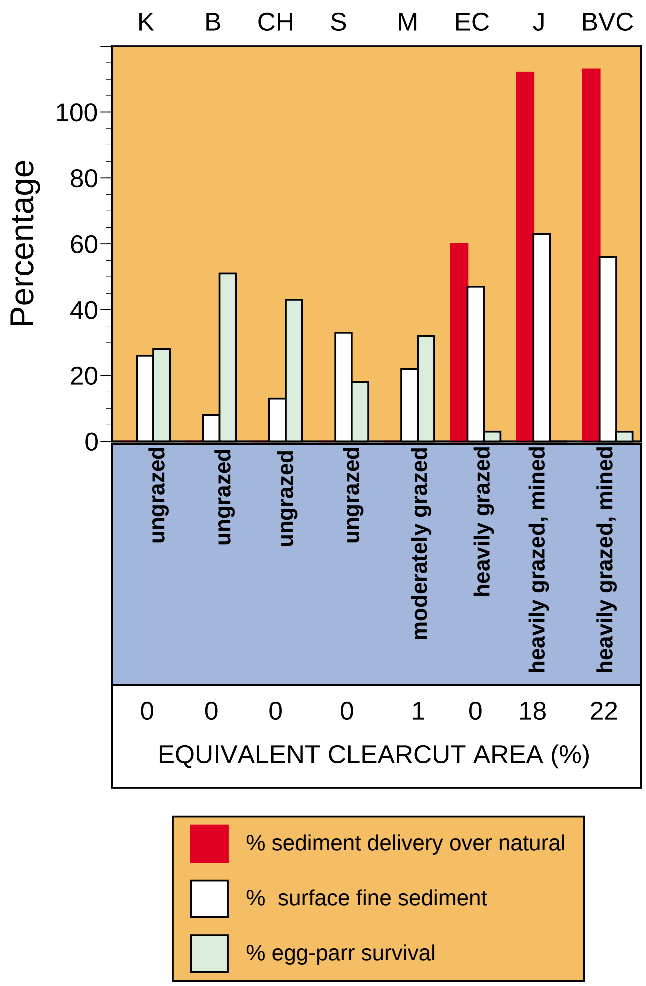

Figure 5. Estimated sediment delivery, surface fine sediment, and estimated egg to parr salmon survival in watersheds with varying levels of estimated Equivalent Clearcut Area (ECA). All streams are tributary to the South and Middle Fork of the Salmon River and located within the Idaho batholith. Egg to parr salmon survival estimated from regression of survival vs. fine sediment from the data of Scully and Petrosky (1991). Fine sediment data and estimated sediment delivery are from Rich et al. (1992), Boise National Forest (1993), Challis National Forest (1993), and NMFS (1993). Ungrazed watersheds with ECA=0 are assumed to have sediment delivery at 0% over natural. Where ECA levels were given as a range, the median of the range was used. Abbreviations are as follows: K=Knapp Creek; B=Beaver Creek; CH=Chamberlain Creek; S=Sulfur Creek; M=Marsh Creek; EC=Elk Creek; J=Johnson Creek; BVC=Bear Valley Creek. In heavily grazed EC, substrate conditions are degraded and salmon survival depressed at ECA=0. Salmon survival has been reduced by high levels of fine sediment, especially in streams subjected to the combined impacts of grazing, logging, and mining. Egg to parr survival in Johnson Creek is estimated to be about 0.5% (NMFS, 1993). Elevated sediment delivery from combined sources is the primary cause of degraded substrate conditions and reduced salmon survival.

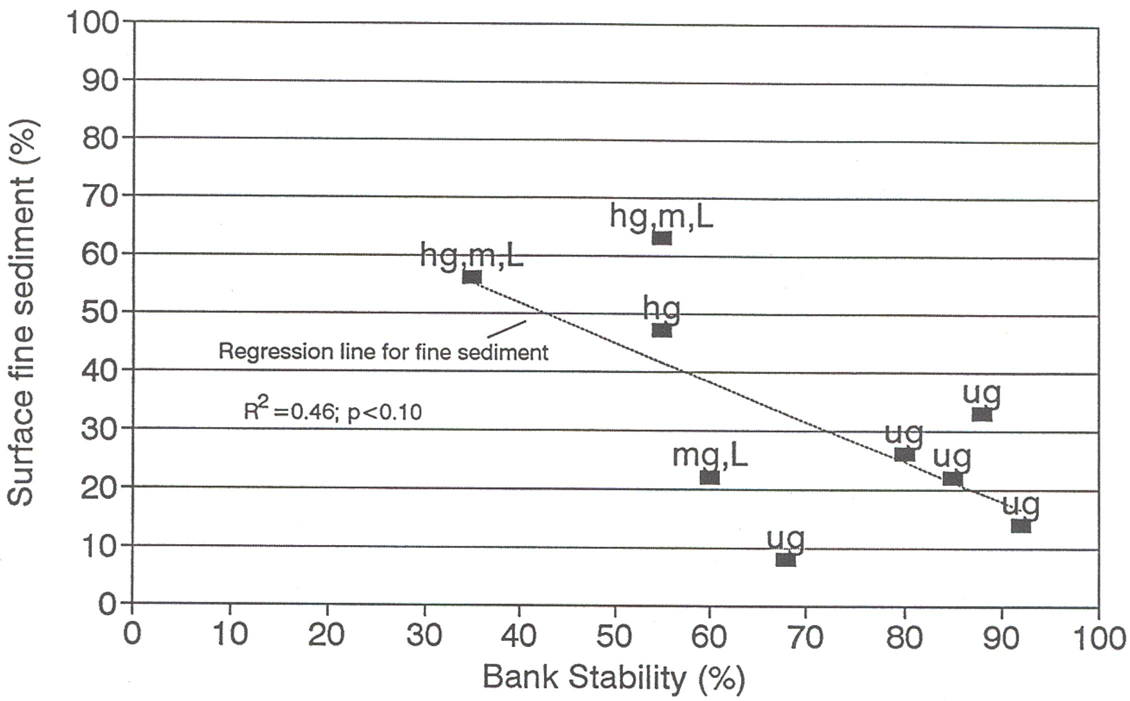

Figure 6. Percent surface fine sediment and bank stability in tributaries to the Middle Fork and South Fork of the Salmon River with differing types of land use. Abbreviations are as follows: ug=ungrazed; mg=moderately grazed; hg=heavily grazed; m=mined; L=logged and roaded. Data are from Johnson, Bear Valley, Porter, Elk, Marsh, Beaver, Sulfur, Knapp, and Cape Horn Creeks on the Boise and Challis National Forest (Rich et al., 1992; Boise National Forest, 1993; NMFS, 1993; Challis National Forest, 1993). While surface fine sediment levels are inversely correlated to bank stability ($p < 0.05$), watershed scale sediment delivery from all sources is probably a major factor in fine sediment levels. Fine sediment levels are highest in streams in watersheds that have been grazed, logged, and mined. This is probably due to sediment delivery from these combined sources. Salmon survival decreases with increasing fine sediment levels (See Figures 1 and 5).

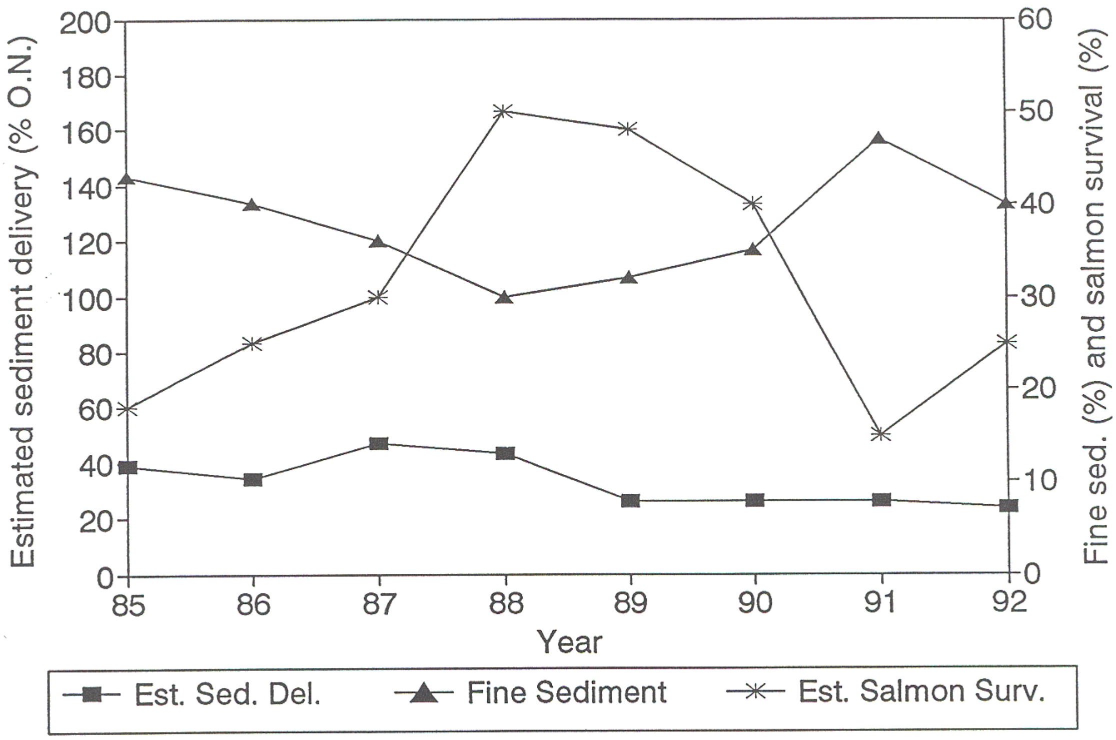

Figure 7. Trends in estimated sediment delivery (Clearwater National Forest (CNF), unpublished WATBAL outputs), percent fine sediment by depth (CNF, 1993), and estimated salmon survival-to-emergence (STE) as a function of fine sediment levels (USFS, 1983) in Pete King Creek, a logged and roaded watershed in the Idaho batholith on the CNF. The stream has also been subjected to some grazing and mining. Active removal of sediment from in-channel sediment traps has occurred annually upstream of the fine sediment sampling area since 1989. Although data indicate some short-term cyclic oscillations in percent fine sediment by depth, substrate conditions show no statistically significant change from 1985-92 (R^2=0.004; p >> 0.10; slope=0.1 percent fine sediment/yr) under existing sediment loads, even with sediment removal efforts. It is estimated that salmon STE remains considerably depressed due to high levels of fine sediment. It appears that sediment loads remain at levels that prevent initiation of sustained recovery in substrate conditions. Sediment delivery is probably underestimated because CNF sediment models ignore contributions from roads more than six years old even though roads of this vintage continue to erode at rates far above natural. It is estimated that the Equivalent Clearcut Area ranged from about 13-16% of the watershed area from 1985-1992 (CNF, unpublished WATBAL outputs).

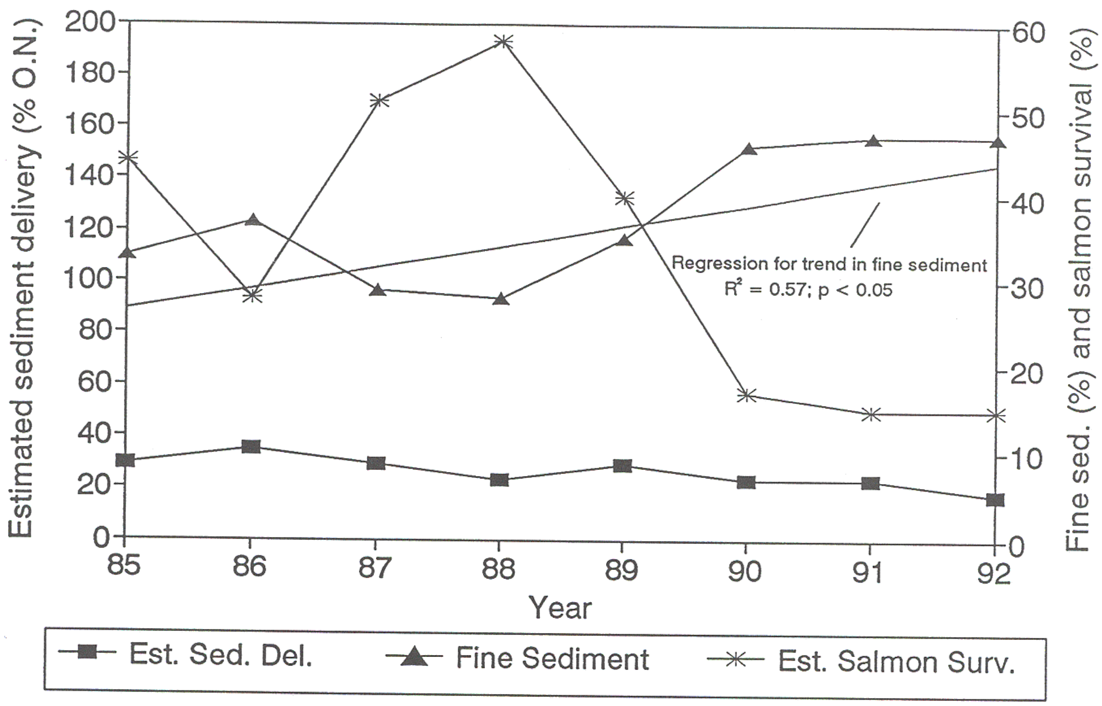

Figure 8. Trends in estimated sediment delivery (Clearwater National Forest (CNF), unpublished WATBAL outputs), fine sediment by depth (CNF, 1993), and estimated salmon survival-to-emergence (STE) as a function of fine sediment levels (USFS, 1983) in Deadman Creek, a logged and roaded watershed in the Idaho batholith on the CNF. Some mining has also occurred in the watershed. Although data indicate some short-term cyclic oscillations in fine sediment conditions, there was a statistically significant ($p < 0.10$) increase in percent fine sediment by depth (R^2=0.57; slope=2.45 percent fine sediment/yr) over the entire time period (1985-1992) despite declines in sediment delivery, indicating that sediment delivery is still in excess of the ability of the stream to transport sediment. Based on the increases in fine sediment it estimated that salmon STE has also deteriorated during the time period and remains low. Improvement in substrate conditions and salmon survival will require further reductions in sediment loading. Estimated sediment delivery is probably underestimated. CNF sediment models ignore contributions from roads more than six years old even though roads of this vintage continue to erode at rates far above natural. It is estimated that the Equivalent Clearcut Area ranged from about 4-9% of the watershed area from 1985-1992 (CNF, unpublished WATBAL outputs).

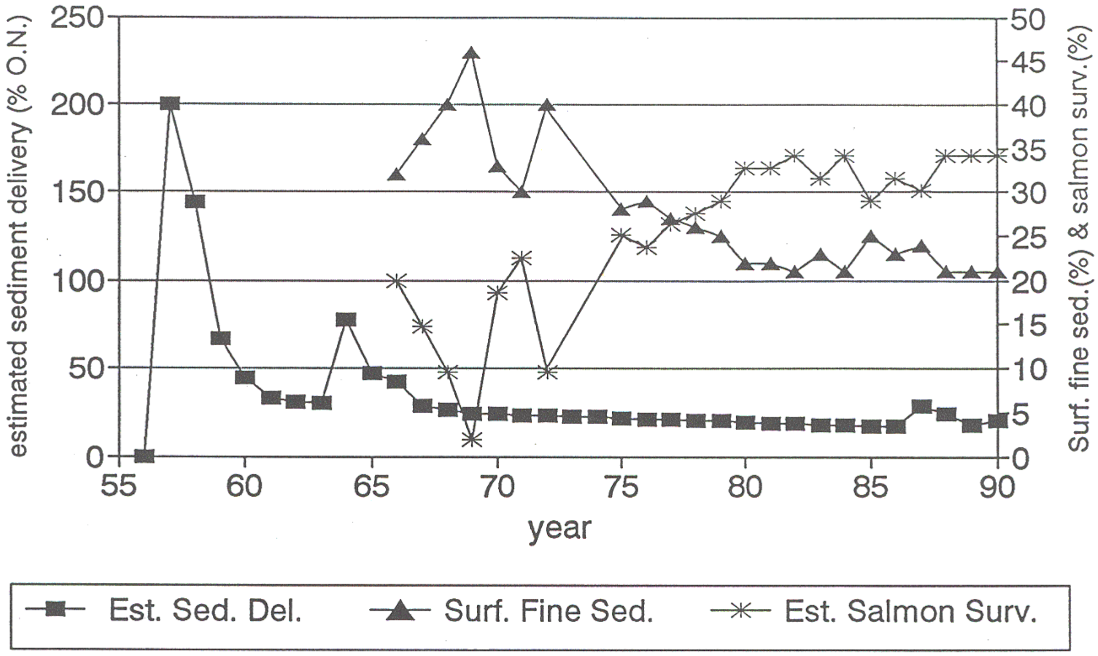

Figure 9. Trends in percent surface fine sediment (Platts et al., 1989), estimated sediment delivery in Dollar Creek (Megahan et al., 1992), and estimated egg to parr salmon survival South Fork of the Salmon River (SFSR). Estimated sediment delivery from Dollar Creek, a subwatershed of the SFSR, is only as an index of estimated sediment delivery in the SFSR because Dollar Creek has had a slightly different land-use and fire history (D. Burns, Payette National Forest Fish. Bio., pers. comm., 1994). Salmon survival is estimated as a function of fine sediment based on a linear regression analysis of the data of Scully and Petrosky (1991). The recovery of substrate conditions in the SFSR was initially rapid after the catastrophic sedimentation, but ceased after sediment delivery was stabilized at about 15-20% over natural (Idaho Dept. of Health and Welfare, 1991). It appears that substrate conditions are now in equilibrium with existing sediment loads; recovery will not resume unless sediment delivery is reduced further (Platts et al., 1989; Idaho Dept. of Health and Welfare, 1991). The catastrophic sedimentation of the SFSR reduced the endemic salmon populations by significantly reducing salmon survival (Platts et al., 1989). Spawner-recruit data independently indicate that habitat productivity in the SFSR has been significantly reduced by the persistent effects of habitat degradation (Schaller and Petrosky, *in press*). Salmon survival is estimated to have increased about ten-fold since reaching a nadir in 1969.

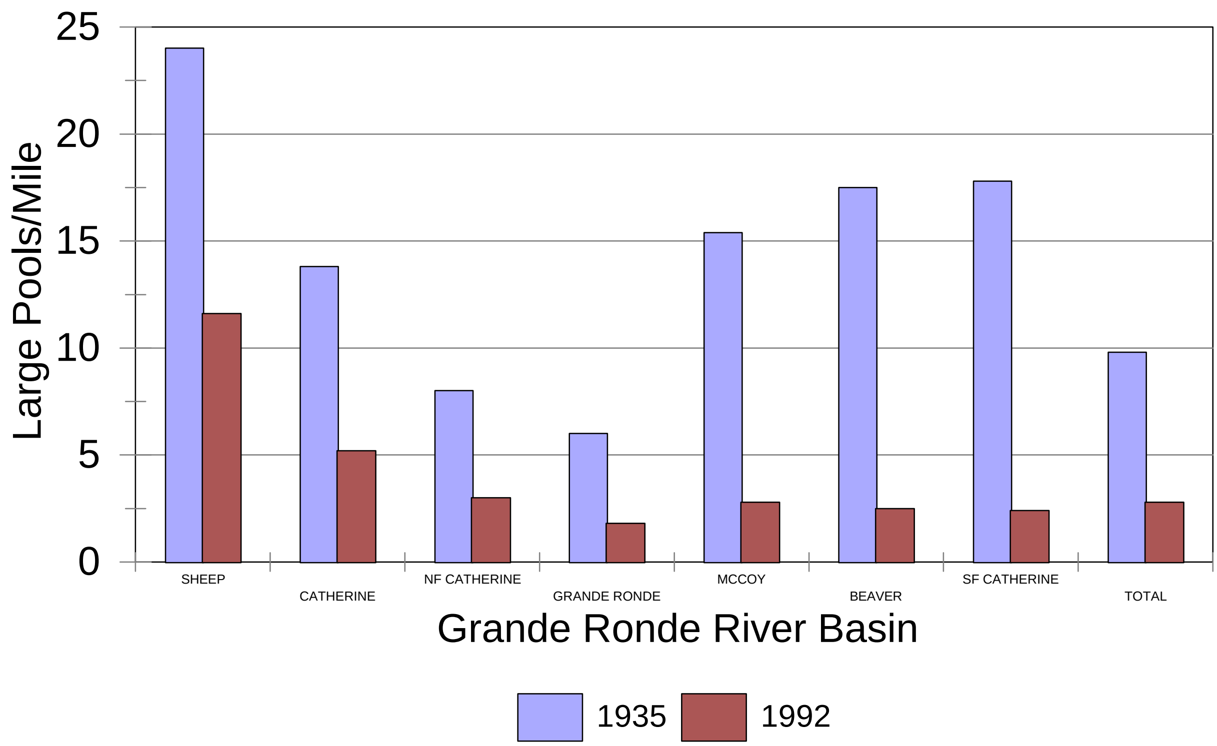

Figure 10. Frequency of large pools in managed watersheds in the Grande Ronde River Basin between 1935 and 1992. Large pools defined as having surface area greater than about 215 ft^2 and a depth greater than about 3.3 ft (McIntosh et al., 1994). Although the Grande Ronde River was affected by the Tanner Gulch fire and flood event in 1989, pool losses were as great in streams unaffected by fire/flood (e.g., Catherine Creek). Causes of pool loss have been attributed to the loss of riparian vegetation, disruption of channel morphology, and sedimentation caused by mining, splash dams, grazing, logging, and a massive program of road construction (McIntosh, 1992; McIntosh et al., 1994). Data from USFS, Pacific Northwest Research Station, unpublished.

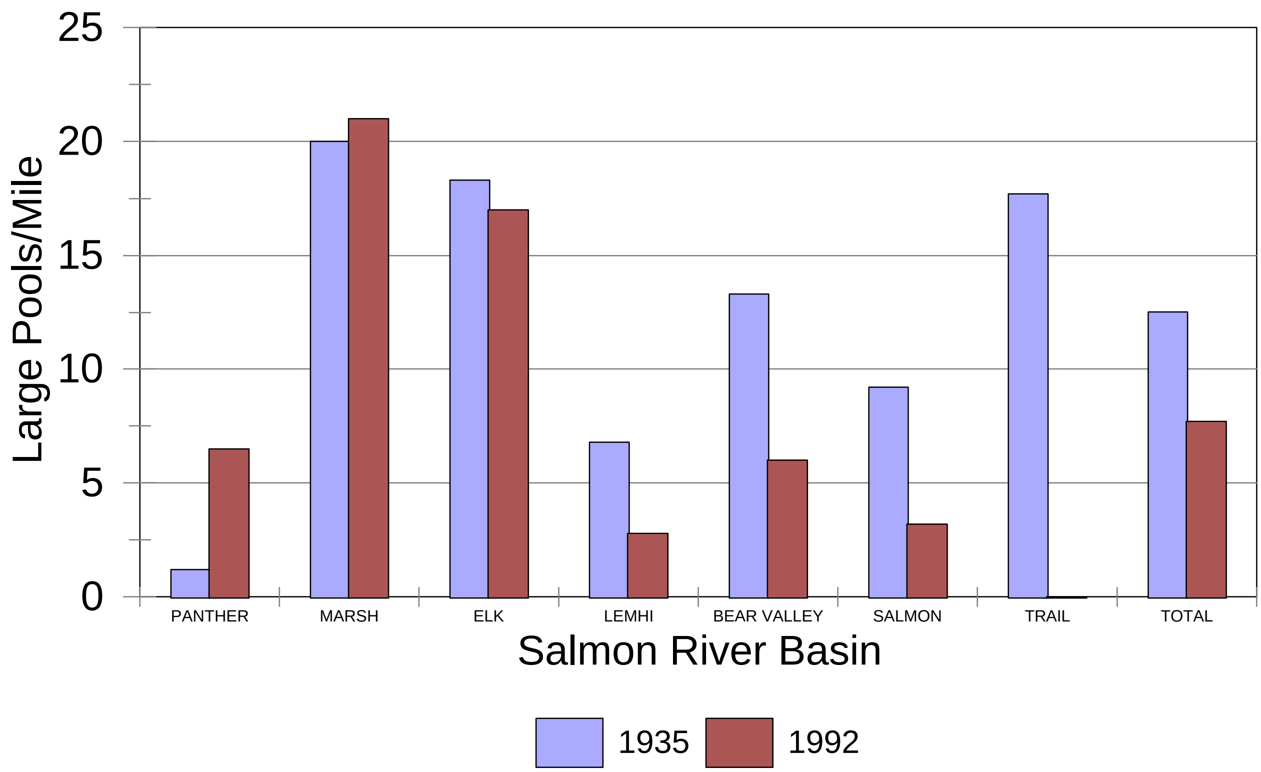

Figure 11. Frequency of large pools in managed watersheds in the Salmon River Basin between 1935 and 1992. Large pools defined as having surface area greater than about 215 ft^2 and a depth greater than about 3.3 ft (McIntosh et al., 1994). All basins have been subject to logging, road construction, and grazing; some have been subjected to mining. Causes of pool loss have been attributed to the loss of riparian vegetation, disruption of channel morphology, and sedimentation caused by mining, grazing, logging, and road construction. In contrast to the managed watersheds, pool frequency in unmanaged watersheds has generally increased slightly, even though some of the unmanaged watersheds have been subjected to fire (See Figure 12). Data from USFS, Pacific Northwest Research Station, unpublished.

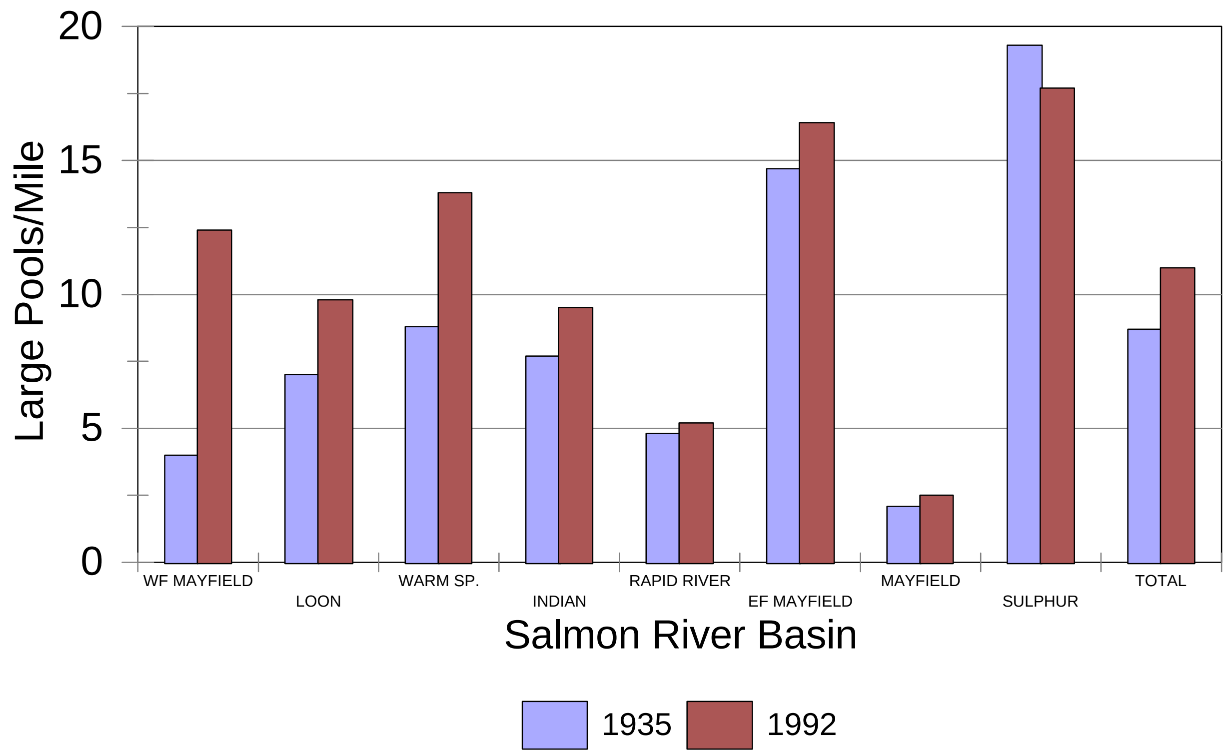

Figure 12. Changes in frequency of large pools (area > 215 ft^2 and depth > 3.3 ft) in unmanaged watersheds in the Middle Fork of the Salmon River Basin from 1935 to 1992. In contrast to the managed watersheds, large pool frequencies in unmanaged watersheds have generally increased slightly, even in Rapid River which was subjected to fire. Data from USFS, Pacific Northwest Research Station, unpublished.

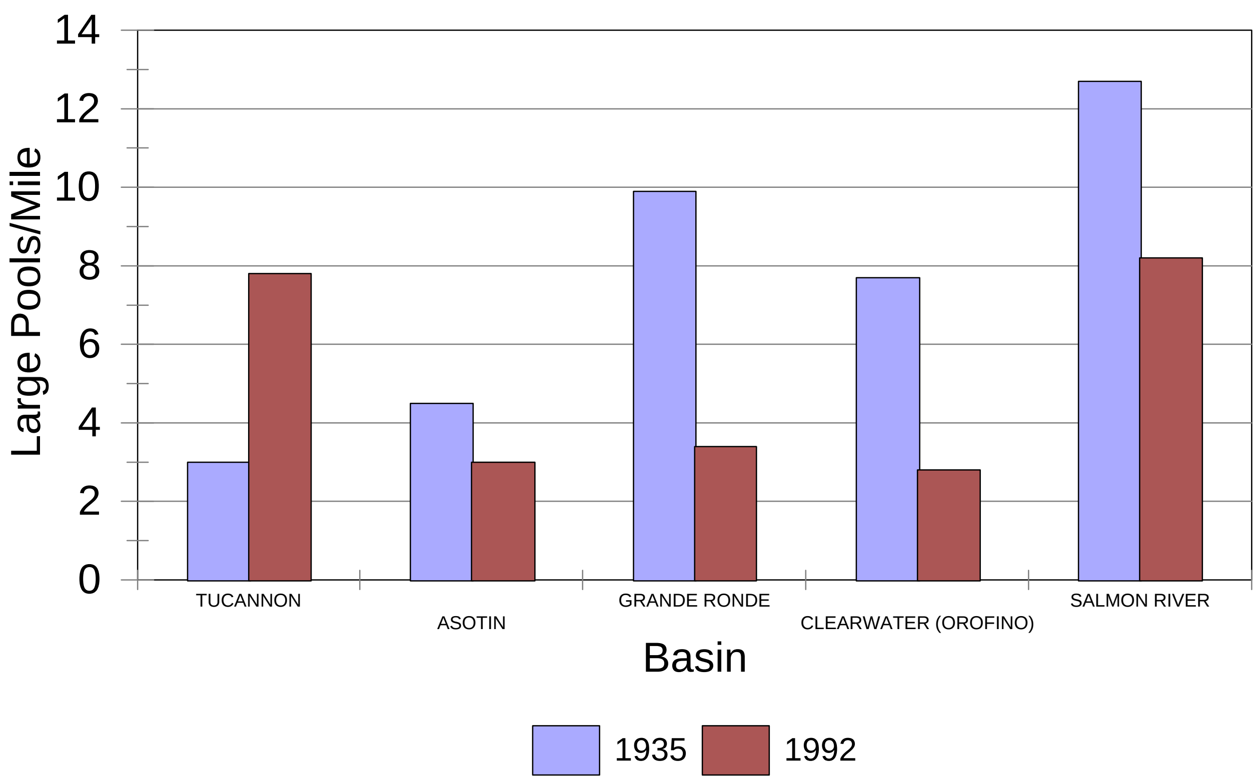

Figure 13. Changes in subbasin scale averages of large pool frequency in managed watersheds in the Snake River Basin from 1935 to 1992. Data from USFS, Pacific Northwest Research Station, unpublished.

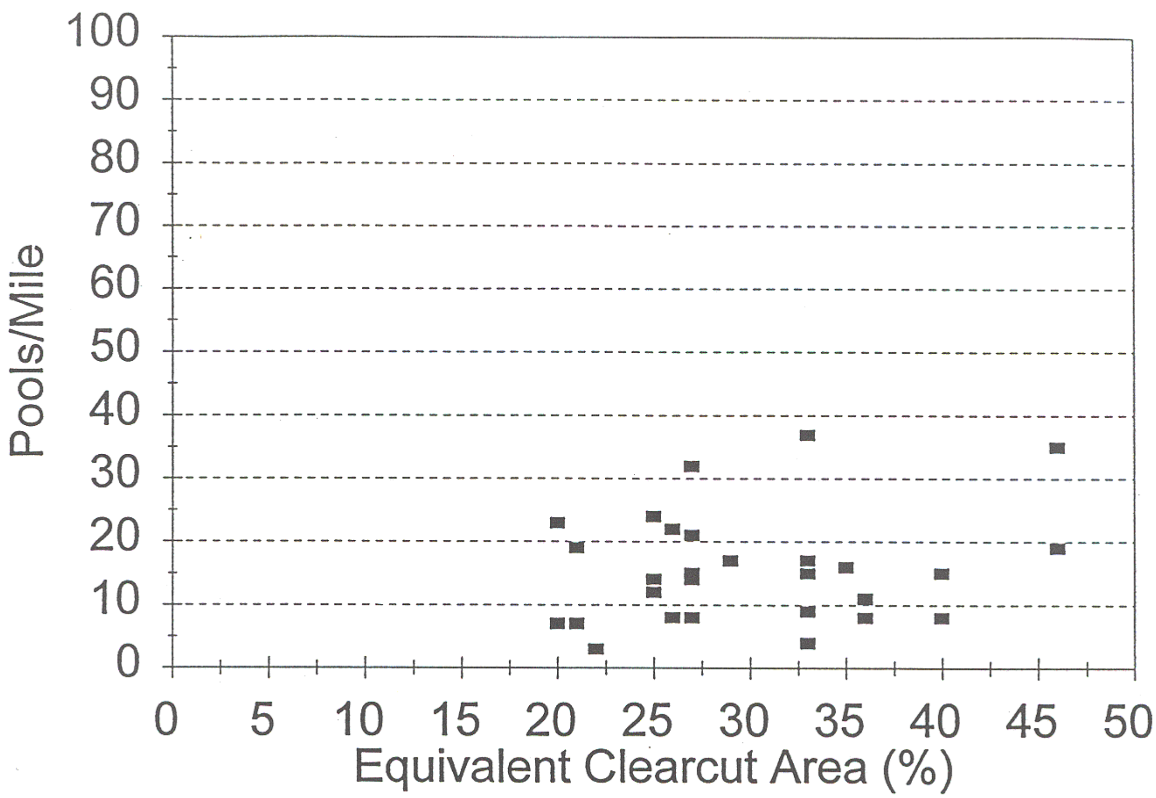

Figure 14. Pool frequency and estimated levels of Equivalent Clearcut Area (ECA) in subwatersheds tributary to the Grande Ronde River on the Wallowa-Whitman National Forest (WWNF) (WWNF, 1992). Pool data are only for fish-bearing portions of the streams within the subwatersheds. Most pools are less than 3 ft deep, so these pool frequency data are not comparable to data in Figures 10-13 for large pools (depth > 3.3 ft). ECA values were estimated based on basal area in logged areas. Roads were treated as equal to clearcuts on a per area basis. All subwatersheds have been subjected to varying levels of grazing; some have been mined. Geology, elevation, area, topography, and vegetation vary among subwatersheds; no attempt was made to stratify data by subwatershed characteristics. Pool frequency is low and pool loss in the Grande Ronde River has been extremely significant from 1935 to 1992 (McIntosh, 1992 (See Figure 10)). Low pool frequency and pool loss have been attributed to the loss of LWD and elevated sedimentation caused by the persistent and combined effects of a massive program of road construction program, grazing, riparian logging, mining, and splash dams (Anderson et al., 1993; McIntosh et al., 1994). Pool frequency is not correlated to the estimated ECA among subwatersheds (R^2=0.03; p >> 0.10).

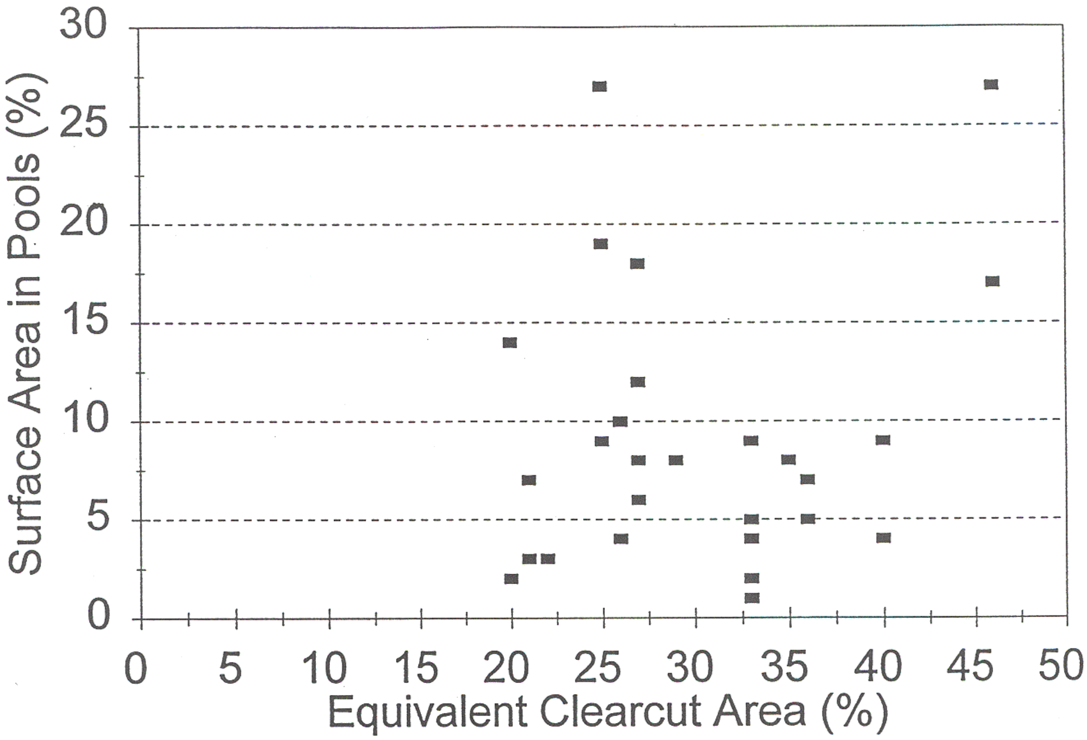

Figure 15. Percent of stream surface area in pools and estimated Equivalent Clearcut Area (ECA) in subwatersheds tributary to the Grande Ronde River on the Wallowa-Whitman National Forest (WWNF) (WWNF, 1992). Pool data are only for fish-bearing portions of streams within the subwatersheds. Most pools are less than 3 ft in depth and data are not comparable to data in Figures 10-13 for large pools (depth > 3.3 ft.). ECA levels were estimated based on basal area in logged areas. Roads were treated as equal to clearcuts on a per area basis. All subwatersheds have been subjected to varying levels of grazing; some have been mined. Geology, elevation, area, topography, and vegetation vary among subwatersheds; no attempt was made to stratify data by subwatershed characteristics. Pool frequency is low and pool loss in the Grande Ronde River has been extremely significant from 1935 to 1992 (McIntosh, 1992). Low pool frequency and pool loss have been attributed to the loss of LWD and elevated sedimentation caused by the persistent and combined effects of a massive road construction program, grazing, riparian logging, mining, and splash dams (Anderson et al., 1993; McIntosh et al., 1994).

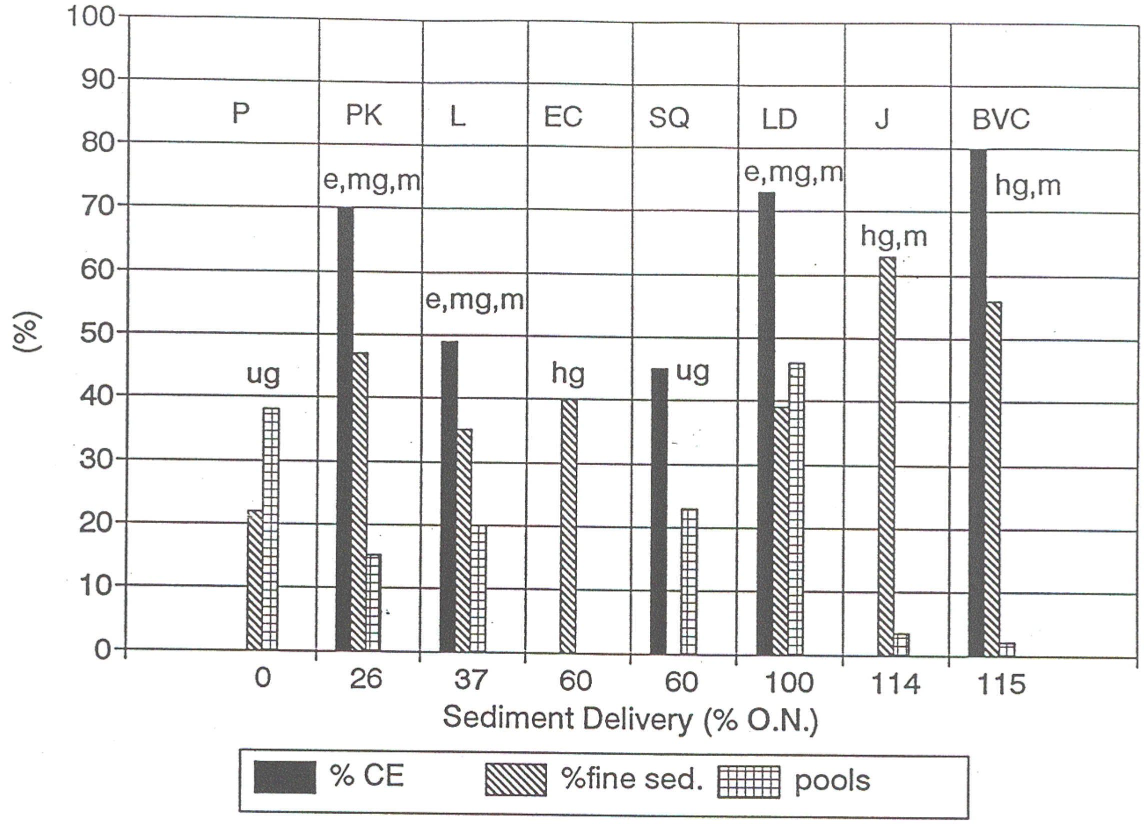

Figure 16. Conditions in salmon habitat in watersheds in the Idaho batholith on the Clearwater (CNF) and Boise National Forests (BNF) at various levels of estimated sediment delivery in percent over natural (% O.N.). Abbreviations in figure are as follows: ug=ungrazed; mg=moderately grazed; hg=heavily grazed; m=mined; e=enhancement efforts have been in place that may have affected the plotted habitat conditions; BVC=Bear Valley Creek (on the BNF); J=Johnson Creek (on the BNF); LD=Eldorado Creek (on the CNF); SQ=Squaw Creek (on the CNF); PK=Pete King Creek (on the CNF); P=Porter Creek (on the BNF); L=Lolo Creek (on the CNF); EC=Elk Creek (on the BNF). (Data Sources: Espinosa and Lee, 1991; CNF, 1991a; CNF, 1992; CNF, 1993; BNF, 1993; NMFS; 1993; CNF, unpublished WATBAL runs). Cobble embeddedness (CE) data unavailable for P, EC, SQ, and J. Pool data unavailable for EC. All watersheds <u>except</u> P and EC have been logged and roaded. Data generally indicate that habitat conditions deteriorate at increasing levels of estimated sediment delivery. Estimated sediment delivery explains more of the variability in cobble embeddedness and percent fine sediment by depth than ECA levels, though none of correlations were statistically significant ($p > 0.10$) (See Table 3). PK, LD, and L do not have pool frequencies that fully reflect the legacy of land management because all three streams have had large woody debris (LWD) added to form pools. It is not known how pool frequency has been altered by LWD additions. Substrate conditions in PK also may not reflect the effects of land management because active sediment trapping and removal has occurred since 1989.

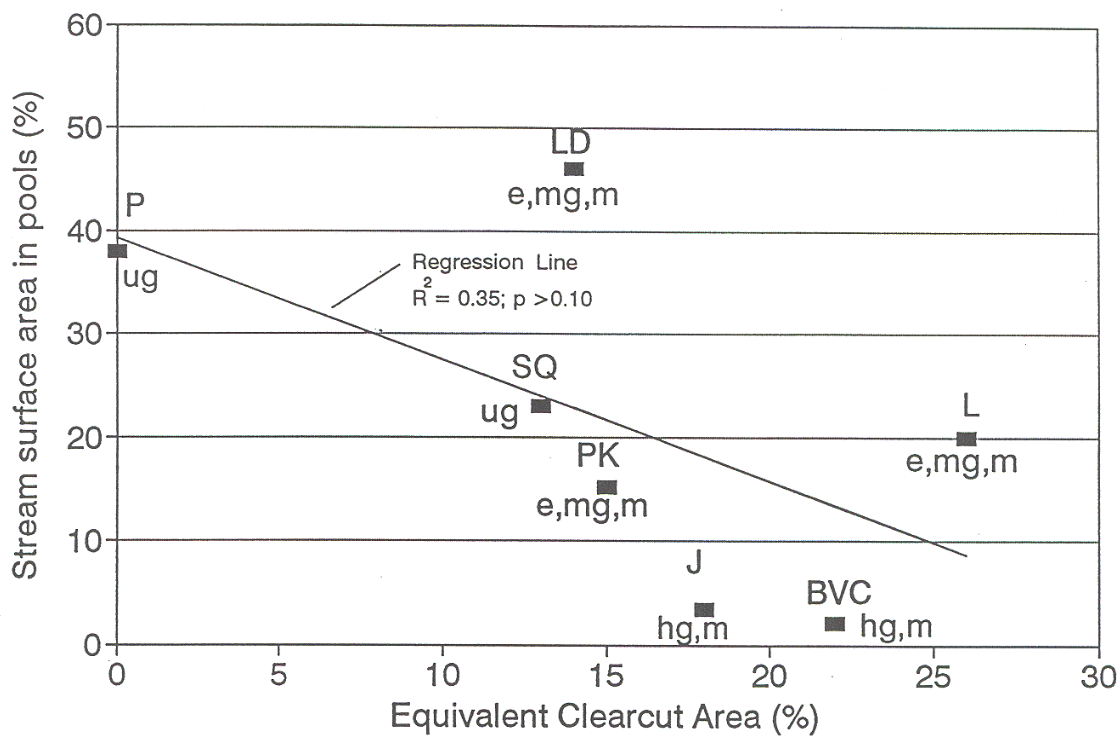

Figure 17. Percent stream surface area in pools and estimated Equivalent Clearcut Area (ECA) in watersheds in the Idaho batholith on the Clearwater (CNF) and Boise National Forests (BNF). Where ECA levels were given as a range, the median within the range was used. Abbreviations in the figure are as follows: ug=ungrazed; hg=heavily grazed; m=mined; e=enhancement efforts have been in place that may have affected the plotted variable; BVC=Bear Valley Creek (BNF); SQ=Squaw Creek (CNF); J=Johnson Creek (BNF); LD=Eldorado Creek (CNF); PK=Pete King Creek (CNF); P=Porter Creek (BNF); L=Lolo Creek (CNF); EC=Elk Creek (BNF). (Data Sources: Espinosa and Lee, 1991; CNF, 1992; BNF, 1993; NMFS; 1993; CNF, unpublished WATBAL runs). Pool frequency tends to decrease with increasing percent ECA among watersheds (R^2=0.35), but correlation is not statistically significant ($p > 0.10$). Pool frequency is lowest in watersheds that have also been subjected to heavy grazing and mining. Bear Valley and Johnson Creeks have low bank stability and high sediment loads caused by grazing (BNF, 1992; NMFS, 1993). PK, LD, and L probably do not have pool frequencies that fully reflect the legacy of land management because all had large woody debris (LWD) added. It is not known to what extent pool frequency has been altered by the LWD additions.

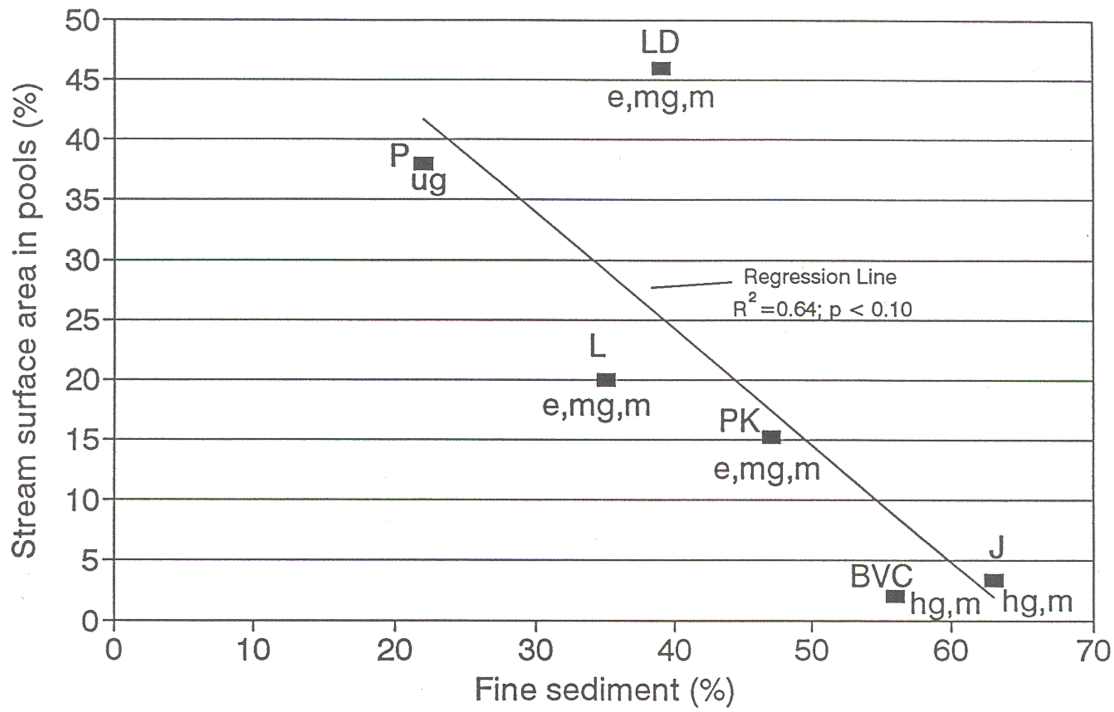

Figure 18. Percent of stream surface area in pools and percent fine sediment in streams in watersheds in the Idaho batholith on the Clearwater (CNF) and Boise National Forests (BNF). Abbreviations are as follows: ug=ungrazed; mg=moderately grazed; hg=heavily grazed; m=mined; e=enhancement efforts have been in place that may have affected the plotted variable; BVC=Bear Valley Creek (on the BNF); J=Johnson Creek; LD=Eldorado Creek (on the CNF); PK=Pete King Creek (on the CNF); P=Porter Creek (on the BNF); L=Lolo Creek; EC=Elk Creek (on the BNF). (Data from Espinosa and Lee, 1991; CNF, 1992; CNF, 1993; BNF, 1993; NMFS; 1993; CNF, unpublished WATBAL runs). Inverse correlation of percent of stream surface area in pools with percent fine sediment is statistically significant (R^2=0.64; $p < 0.10$). PK, LD, and L may not have pool frequencies that fully reflect the legacy of land management because all three streams have had large woody debris (LWD) added to form pools. It is not known to what extent pool numbers have been altered by the LWD additions.

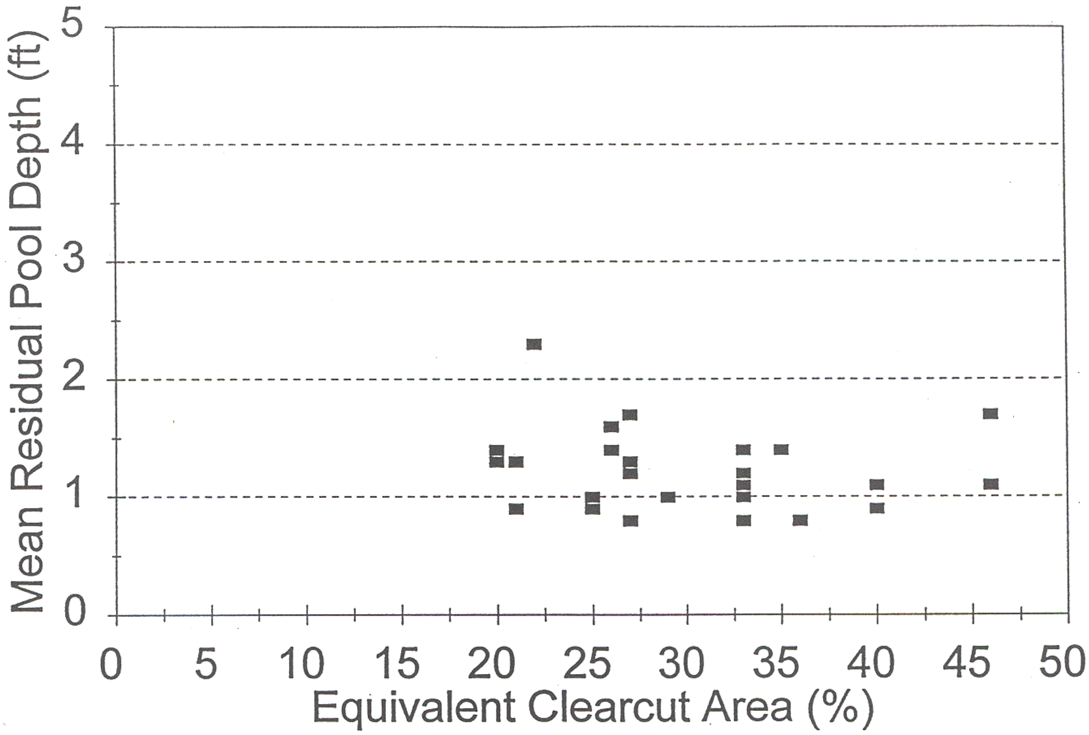

Figure 19. Mean residual pool depth and estimated Equivalent Clearcut Area (ECA) within subwatersheds tributary to the Grande Ronde River on the Wallowa-Whitman National Forest (WWNF) (WWNF, 1992). Pool depth data are only for fish-bearing portions of the streams within the subwatersheds. ECA levels were estimated based on basal area in logged areas. Roads were treated as equal to clearcuts on a per area basis. All subwatersheds have been subjected to varying levels of grazing; some have been mined. Geology, elevation, area, topography, and vegetation vary among subwatersheds; no attempt was made to stratify data by subwatershed characteristics. Data clearly indicate that large pools (> 3 ft deep) are exceedingly rare. Pool loss in the Grande Ronde River has been considerable (>50%) over the past 50 years (McIntosh, 1992). Low pool frequency and pool loss have been attributed to the loss of LWD combined with elevated sedimentation caused by the combined effects of a massive road construction program, grazing, riparian logging, mining, and splash dams (McIntosh, 1992; McIntosh et al., 1994). These same factors may be have contributed to the lack of large pools. Mean residual pool depth is uncorrelated to estimated ECA among subwatersheds (R^2=0.03; $p > 0.10$).

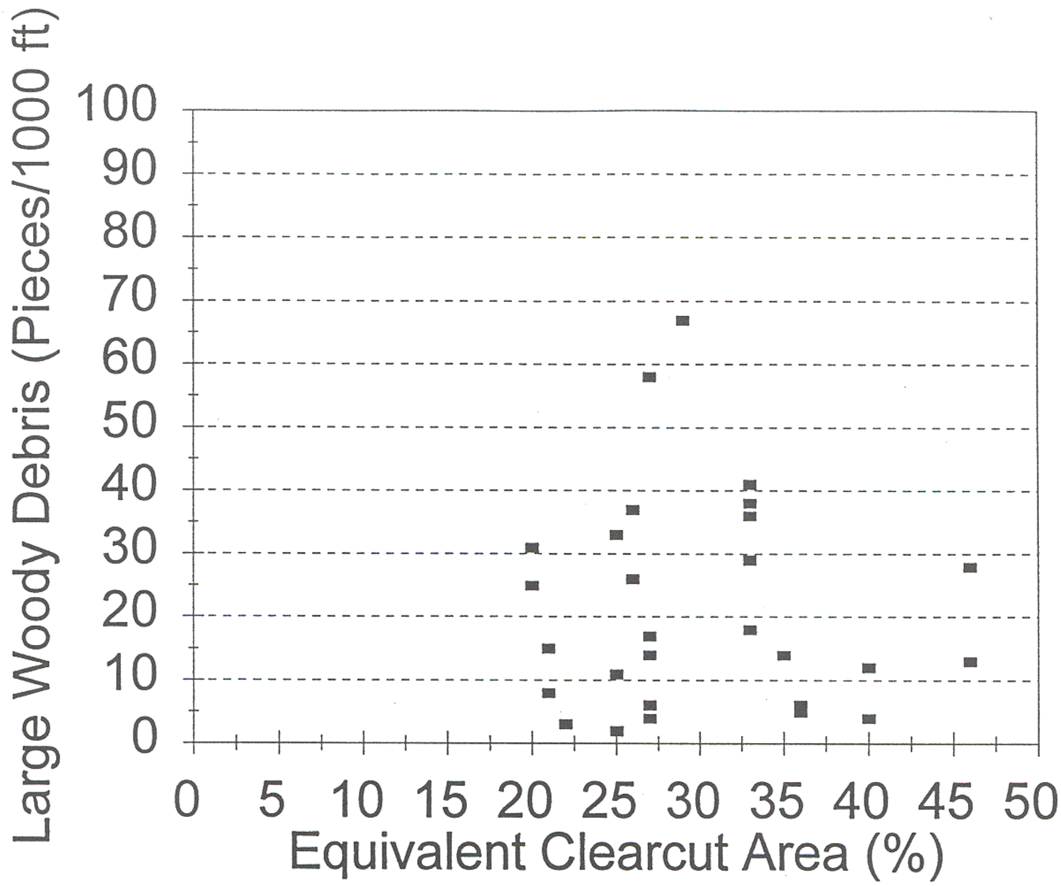

Figure 20. Frequency of large woody debris (LWD) and estimated Equivalent Clearcut Area (ECA) within subwatersheds tributary to the Grande Ronde River on the Wallowa-Whitman National Forest (WWNF) (WWNF, 1992). LWD defined as pieces with mean diameter greater than 1 ft and length greater than 35 ft. LWD data are only for fish-bearing portions of the streams within the subwatersheds. ECA levels were estimated based on basal area within logged areas. Roads were treated as equal to clearcuts on a per area basis. All subwatersheds have been subjected to varying levels of grazing; some have been mined. Geology, elevation, area, topography, and vegetation vary among subwatersheds; no attempt was made to stratify data by subwatershed characteristics. Minimum LWD targets have been set for the streams at 20 pieces of LWD/1000 ft (Anderson et al., 1993). However, field evaluations indicate that LWD frequency in some undisturbed streams in the Blue Mountains ranges from 35-60 pieces/1000 ft (J. Rhodes, unpublished field notes, 1993). LWD data shown in the figure do not completely reflect the legacy of land management; LWD has been artificially introduced in many of the tributaries. Nonetheless, LWD frequency is low in many streams and may be lower in the future. Sources of continued LWD recruitment are lacking (Anderson et al., 1992). Pool loss in the Grande Ronde River has been extremely significant over the past 50 years (McIntosh, 1992) and has been partially attributed to the loss of LWD (McIntosh, 1992; Anderson et al., 1993; McIntosh et al., 1994). LWD frequency is not significantly correlated to ECA levels ($R^2<0.01$; $p > 0.10$).

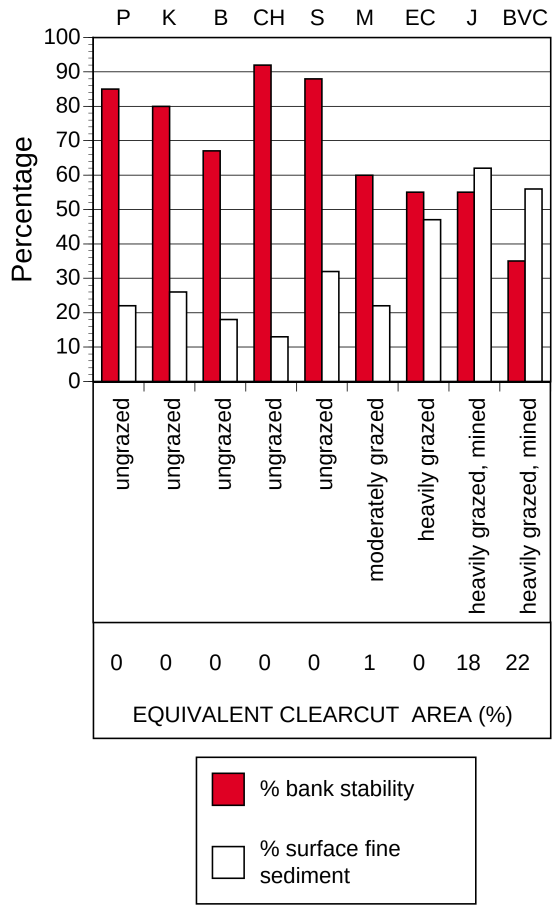

Figure 21. Bank stability, percent surface sand and estimated Equivalent Clearcut Area (ECA) in grazed and ungrazed watersheds in the Idaho batholith. (Data from Rich et al., 1992; Boise National Forest, 1993; Challis National Forest, 1993; NMFS, 1993). Where ECA levels were given as a range, the median was used. Abbreviations are as follows: BVC=Bear Valley Creek; P=Porter Creek; J=Johnson Creek; K=Knapp Creek; EC=Elk Creek; S=Sulfur Creek; CH=Chamberlain Creek; B=Beaver Creek; M=Marsh Creek. EC data clearly indicate that heavily grazed systems can be considerably degraded with respect to bank stability and fine sediment even at ECA=0.

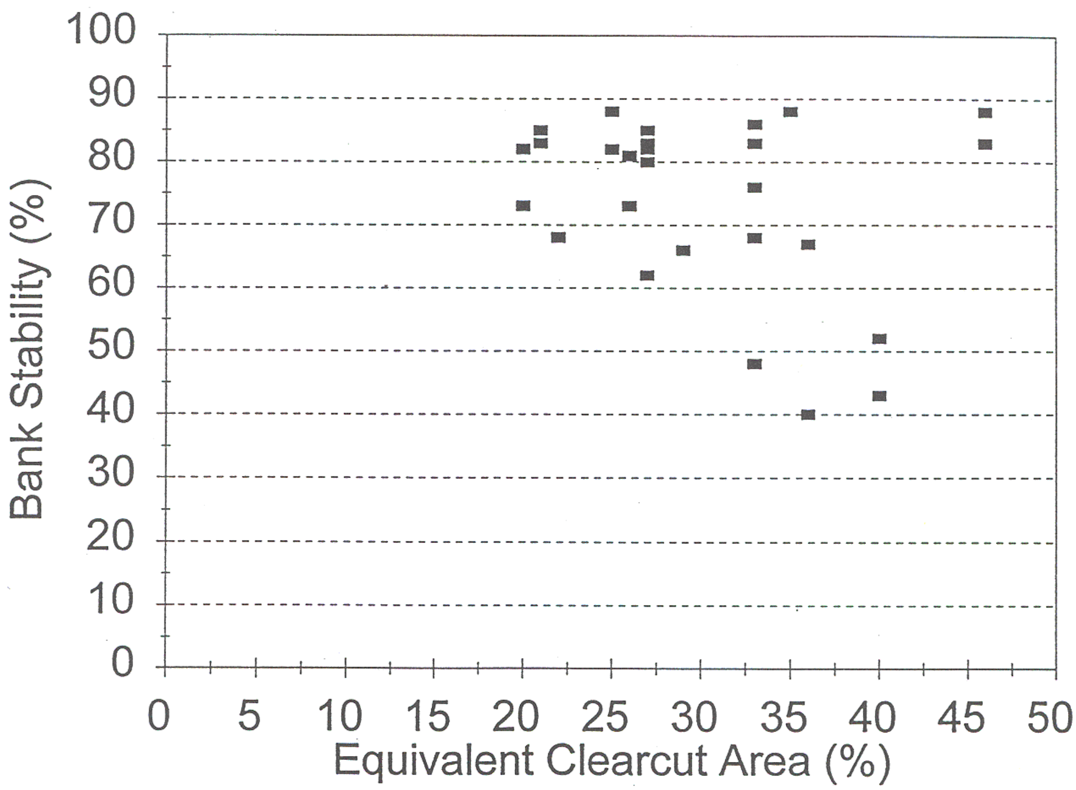

Figure 22. Percent bank stability and estimated Equivalent Clearcut Area (ECA) within subwatersheds tributary to the Grande Ronde River on the Wallowa-Whitman National Forest (WWNF) (WWNF, 1992). Bank stability data are only for fish-bearing portions of the streams within the subwatersheds. ECA levels were estimated based on basal area within logged units. Roads were treated as equal to clearcuts on a per area basis. All subwatersheds have been subjected to varying levels of grazing; some have been mined. Geology, elevation, area, topography, and vegetation vary among subwatersheds; no attempt was made to stratify data by subwatershed characteristics. Bank stability was not significantly correlated to ECA (R^2=0.09; $p > 0.10$).

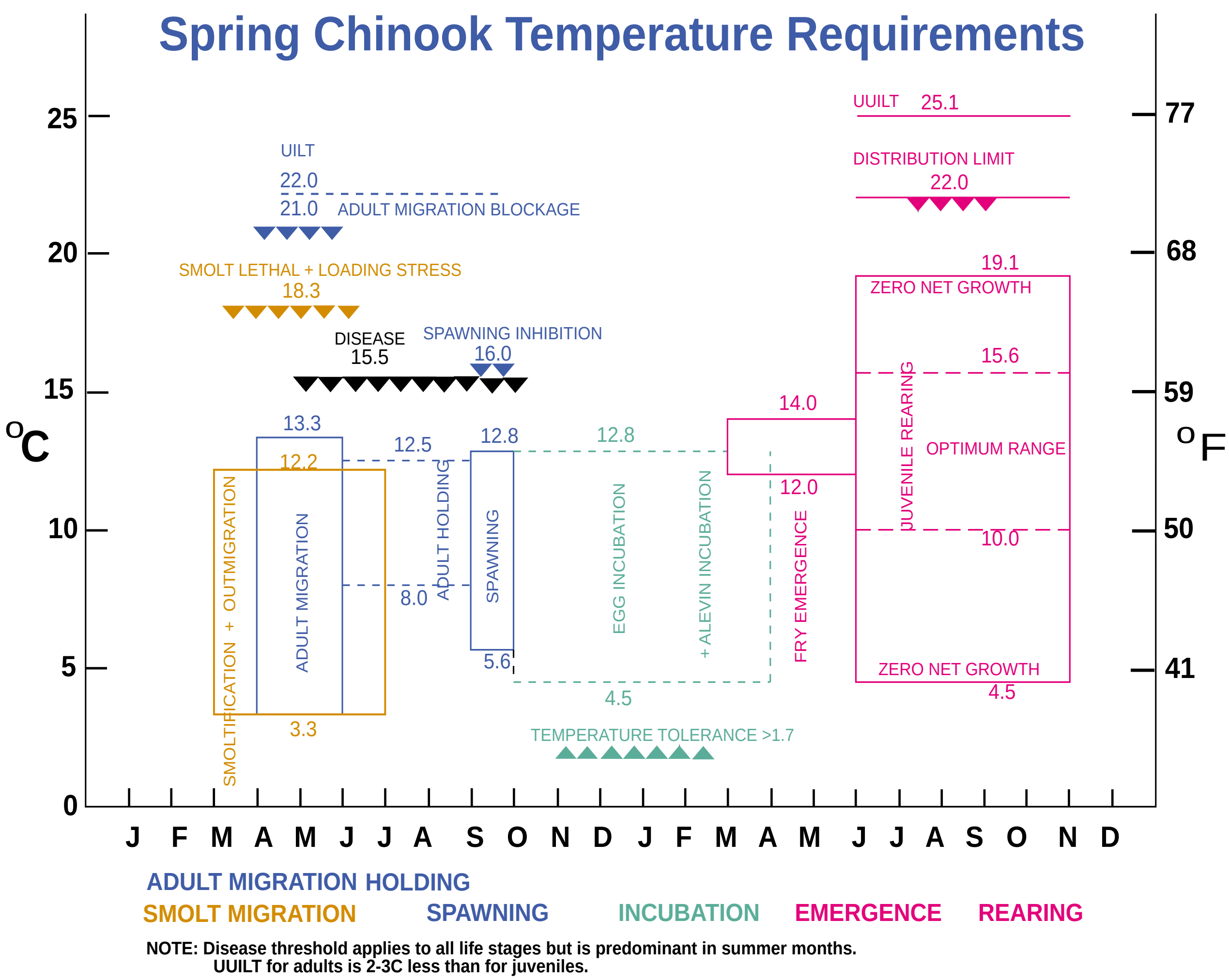

Figure 23. Spring chinook water temperature requirements by lifestage. Although water temperatures in diagram are instantaneous, duration of exposure influences the biologic response of salmon.

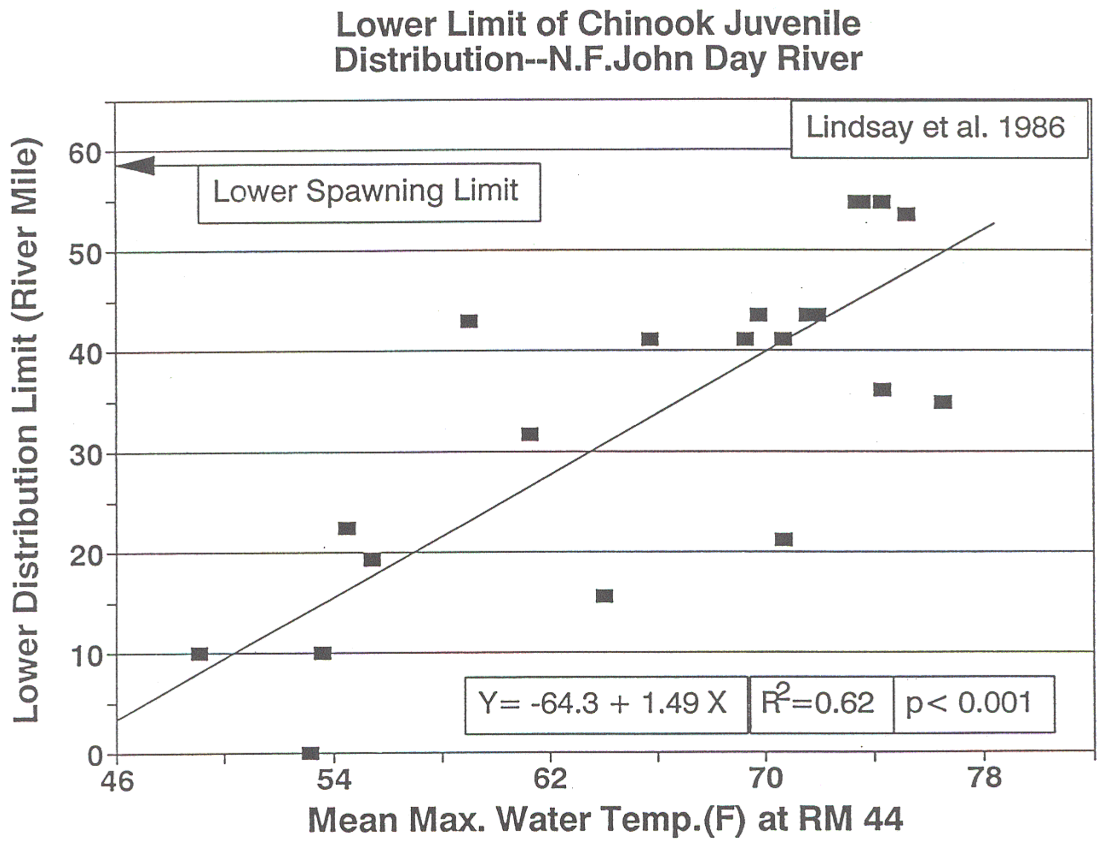

Figure 24. The lower limit of spring chinook juvenile distribution in the North Fork John Day River and mean maximum water temperatures (°F) for 2-week periods preceding juvenile sampling. Water temperatures were measured at about River Mile 44. Redrawn from Lindsay et al. (1986).

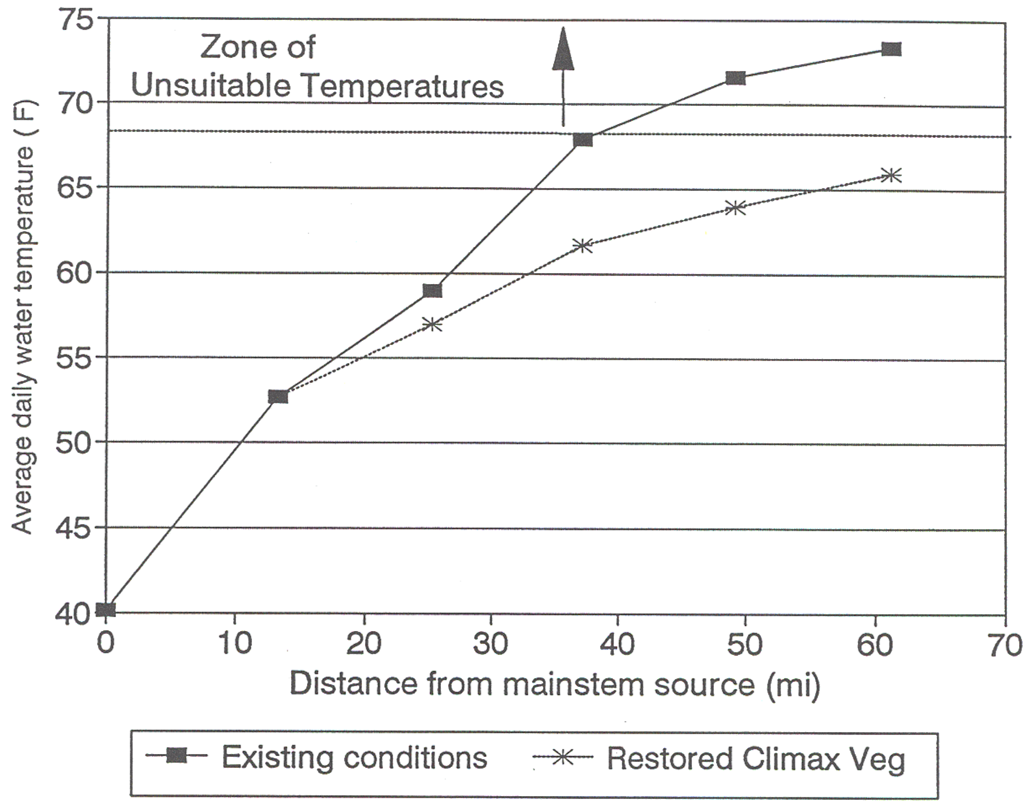

Figure 25. Average daily water temperatures in July under conditions existing on the Tucannon River in Washington in 1985 and modeled average daily water temperatures expected with restoration of riparian vegetation to climax condition. Average daily water temperatures under the restoration of riparian vegetation was modeled via a physically based water temperature model (Theurer et al., 1984) that had been locally calibrated (Theurer et al., 1985). The modeling of water temperature under climax riparian vegetative conditions assumed channel narrowing in response to improved vegetative conditions (Theurer et al., 1985). Average daily water temperatures of greater than 68°F were considered unsuitable for use by chinook salmon (Theurer et al., 1985). Based on model results, restoration of riparian vegetation would increase zone of habitat with suitable mean daily water temperatures by about 24 miles on the Tucannon River (Theurer et al., 1985). After Theurer et al. (1985).

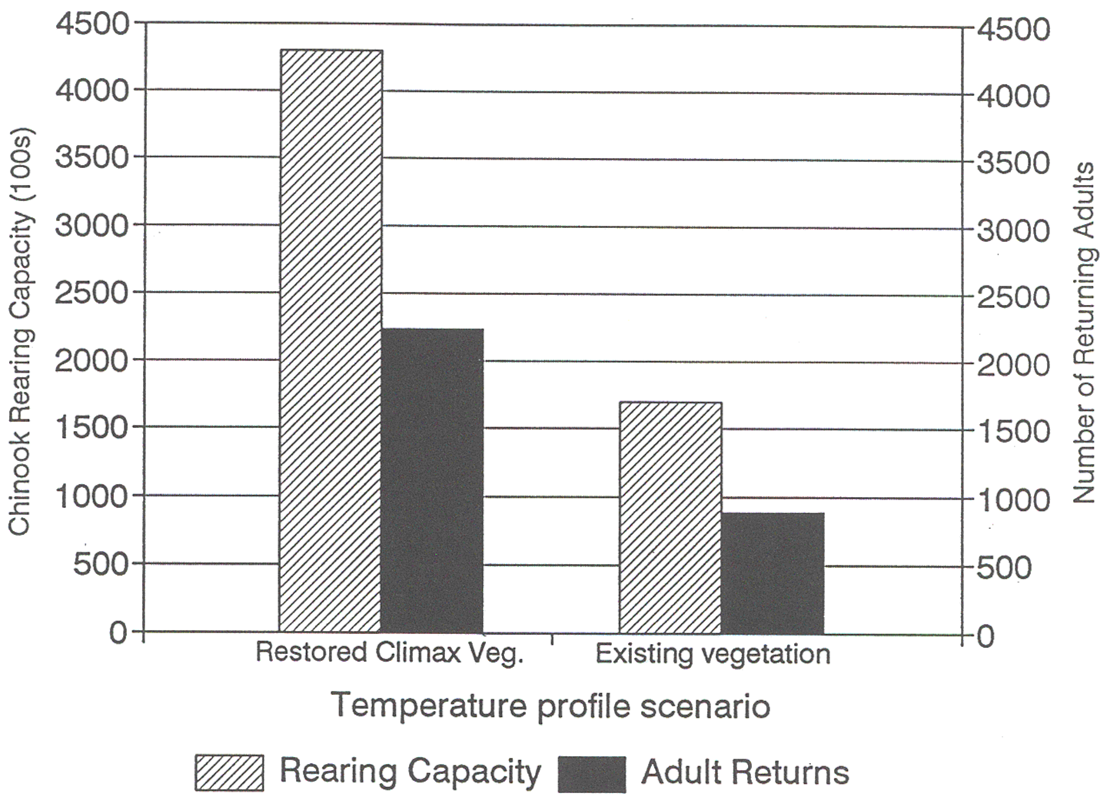

Figure 26. The estimated juvenile rearing capacity and potential number of adult returns with full seeding of rearing capacity for spring chinook salmon in the Tucannon River in eastern Washington under two different water longitudinal water temperature profiles (Theurer et al., 1985). Rearing capacity and potential number of adult returns is higher with restoration of riparian vegetation to climax conditions because it is estimated that this increases the downstream extent of suitable average daily water temperatures (< 68ºF) by about 24 miles (Theurer et al., 1985 (See Figure 25)).

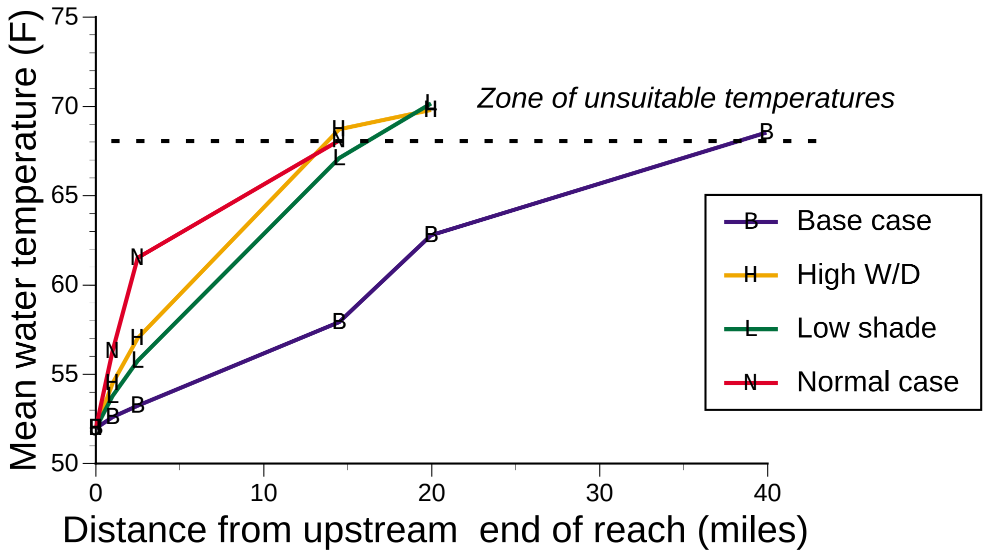

Figure 27. Modeled mean daily water temperature profiles on a small, hypothetical stream under various combinations of stream shading, groundwater inflow, and width-to-depth ratio. All scenarios were modeled via a physically-based water temperature model (Theurer et al., 1984) and meteorological inputs typical of summer conditions in the Blue Mountain Province of Oregon. Tributar streams were not modeled. In all four modeled scenarios, the initial water temperature was 52ºF, the initial discharge was 20 cfs, the temperature of groundwater inflow was 50ºF. Meteorologic inputs were constant among scenarios. The "Base Case" was modeled by setting stream shading = 80%, groundwater inflow = 4 cfs/mi, and width-to-depth ratio = 5.4. The "High W/D" scenario was modeled by setting stream shading = 70%, groundwater inflow = 4 cfs/mi, and width-to-depth ratio = 45. The "Low Shade" scenario was modeled by setting stream shading = 30%, groundwater inflow = 4 cfs/mi, and width-to-depth ratio = 5.4. The "Normal Case" scenario was modeled by setting stream shading = 40%, groundwater inflow = 1 cfs/mi, and width-to-depth ratio = 27. Downstream modeling of water temperatures for each scenario ceased when modeled mean daily water temperatures exceeded 68 ºF. Segments with average water temperatures greater than 68ºF are considered to be unusable by salmon (Theurer et al., 1985). Model results indicate that significant amounts of hospitable rearing area is lost as shade is decreased, width-to-depth ratio is increased, or cool groundwater inputs are decreased.

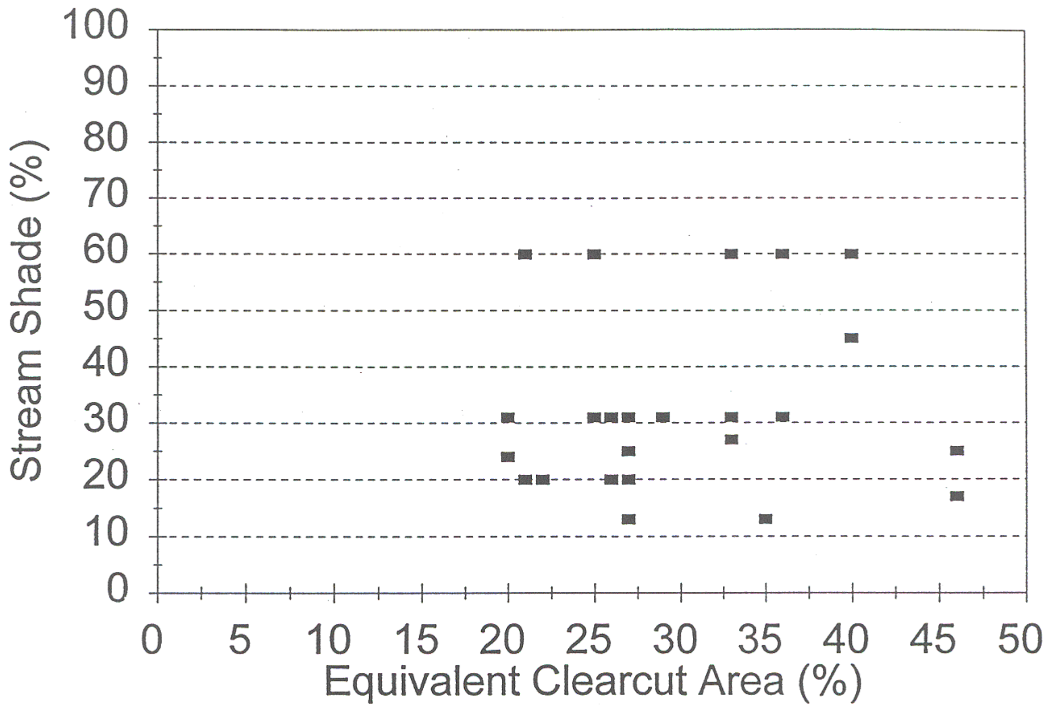

Figure 28. Stream shading and estimated Equivalent Clearcut Area within subwatersheds tributary to the Grande Ronde River on the Wallowa-Whitman National Forest (WWNF) (WWNF, 1992). Stream shading data are only for fish-bearing portions of the streams within the subwatersheds. Plotted shade data are medians of range of shading by shade class. ECA levels were estimated based on basal area within logged units. Roads were treated as equal to clearcuts on a per area basis. All subwatersheds have been subjected to varying levels of grazing; some have been mined. Geology, elevation, area, topography, and vegetation vary among subwatersheds; no attempt was made to stratify data by subwatershed characteristics. Average potential shading for the Grande Ronde River tributaries is estimated to be about 72% (Anderson et al., 1992). Less than 60% stream shade has been considered to represent poor conditions (McCammon, 1993). Data clearly indicate that most streams lack stream shading. These conditions have contributed to elevated water temperatures throughout the Upper Grande Ronde (Anderson et al., 1992 (See Figure 29)). Stream shading has been removed by mining, logging, grazing, and roads in riparian zones within the subwatersheds (Beschta et al., 1991; Anderson et al., 1992; McIntosh et al., 1994).

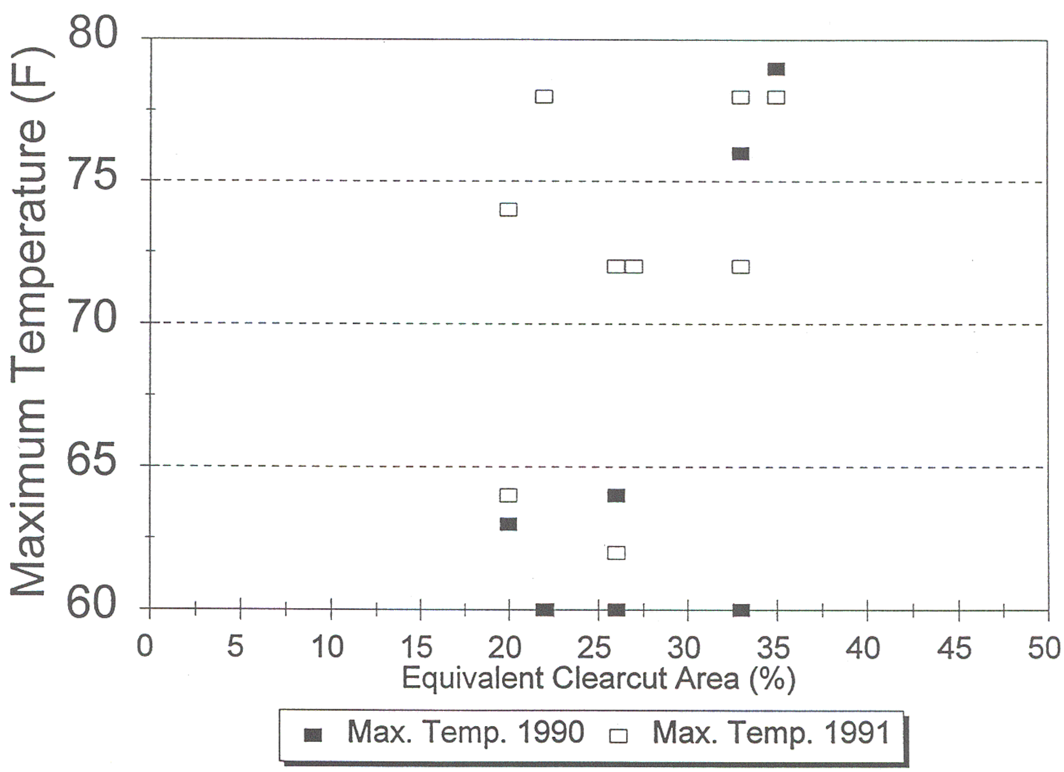

Figure 29. Maximum water temperatures and estimated Equivalent Clearcut Area within subwatersheds tributary to the Grande Ronde River on the Wallowa-Whitman National Forest (WWNF) (WWNF, 1992). ECA levels were estimated based on basal area within logged units. Roads were treated as equal to clearcuts on a per area basis. All subwatersheds have been subjected to varying levels of grazing; some have been mined. Geology, elevation, area, topography, and vegetation vary among subwatersheds; no attempt was made to stratify data by subwatershed characteristics.

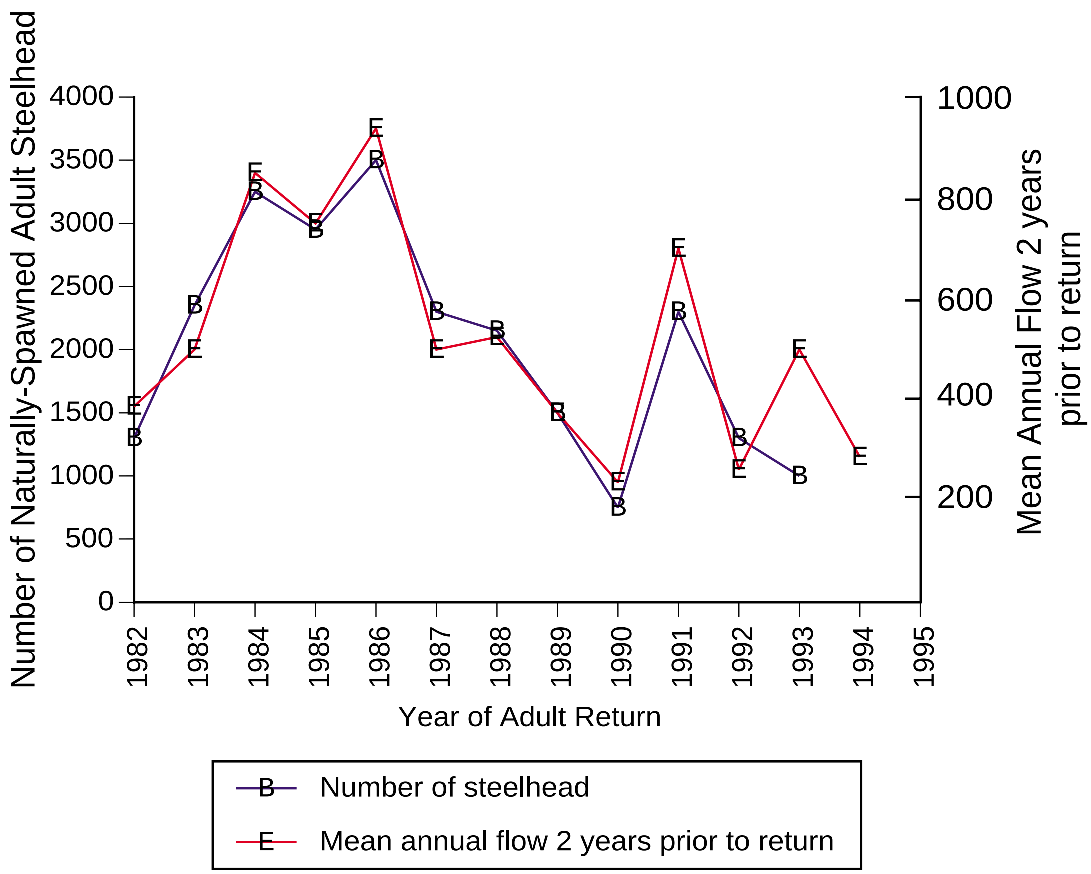

Figure 30. Numbers of natural steelhead returning and mean annual flow two years prior to return year in the Umatilla River in Oregon (Confederated Tribes of the Umatilla Indian Reservation, 1994). Irrigation withdrawals on the Umatilla River significantly reduce flows on the lower river, especially during years with low flows.

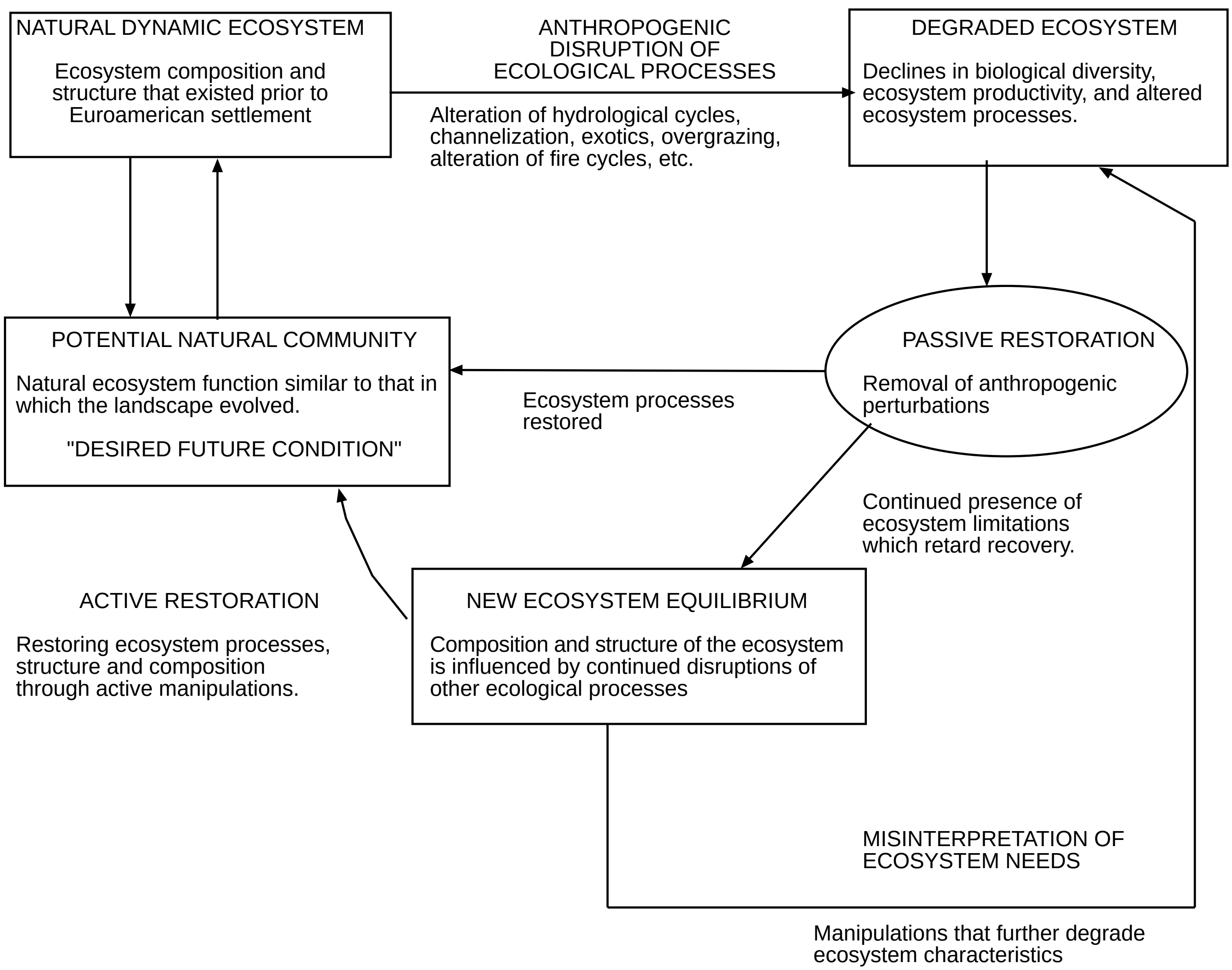

Figure 31. Qualitative, conceptual model for ecosystem response to disturbance and passive and active restoration (From Kauffman et al., 1993).

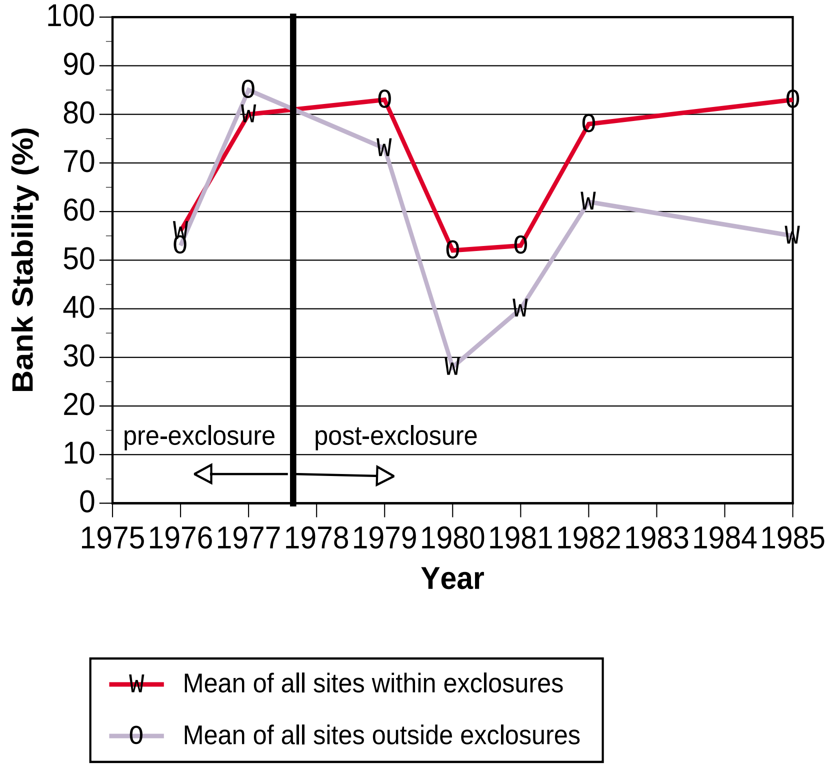

Figure 32. Bank stability trends on Bear Valley Creek on the Boise National Forest (BNF) inside and outside of exclosures. BNF (1993) found that estimated egg-to-parr survival for chinook salmon was positivel correlated with bank stability. Unstable banks contribute to high levels of fine sediment. Salmon survival at the egg-to-parr lifestage is extremely depressed in Bear Valley Creek, averaging about 3% due to high levels of fine sediment (Scully and Petrosky, 1991).

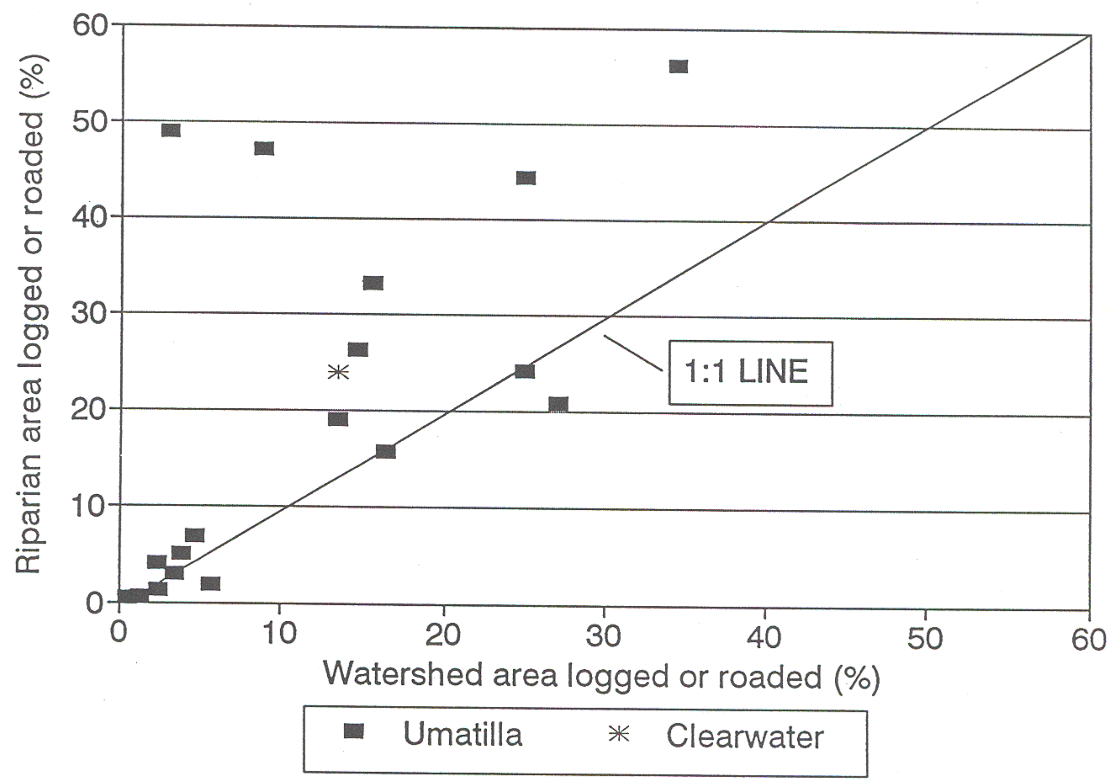

Figure 33. Percent of area logged or roaded in riparian zones and at the subwatershed scale for 19 small subwatersheds tributary to the Tucannon River in the Blue Mountain Province of Washington on the Umatilla National Forest (UNF) and one watershed (Eldorado) tributary to the Middle Fork of the Clearwater River on the Clearwater National Forest (CNF) (CNF, 1992). For UNF data, logged or roaded area includes only clearcuts and shelterwood harvest less than ten years old and roads; selectively harvested areas were ignored (L. Bach, UNF hydrologist, pers. comm., 1993). Selection harvest has occurred in most of the 19 subwatersheds to varying degrees. CNF includes logged units less than 30 years old and roads. The bulk of the data points are above the 1:1 line indicating that the fraction of the riparian area disturbed by logging and roads in these systems is generally greater than the fraction of the watershed disturbed by logging or roads.

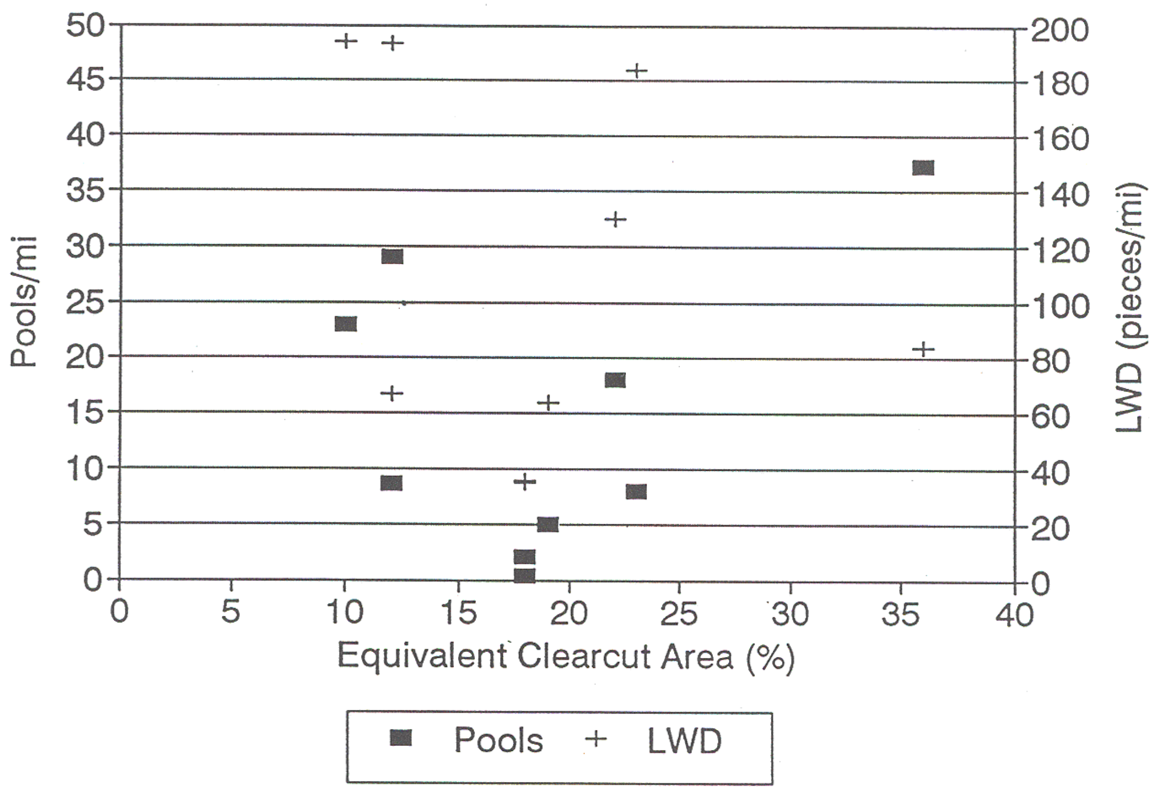

Figure 34. Frequency of large woody debris (LWD), pool frequency, and Equivalent Clearcut Area (ECA) in 9 watersheds tributary to the North Fork of the John Day River in the Blue Mountain Province of Oregon on the Umatilla National Forest (UNF, 1993). ECA values only include shelterwood and clearcut areas less than 10 years old and roads. Selection harvest has been considerable in all 9 watersheds. Livestock grazing occurs in all 9 watersheds. Watershed area, elevation, soils, and vegetation varies among watersheds; no attempt was made to stratify data by watershed attributes. Pool frequencies of less than about 28 pools/mile have been considered to represent poor conditions in these small streams (McCammon, 1993). LWD frequencies of less than about 106 pieces per mile are below the standards recommended for the Upper Grande Ronde River (Anderson et al., 1993); LWD frequencies of less than 10 pieces per mile have been considered to represent poor conditions (McCammon, 1993). Neither LWD frequency nor pool frequency were significantly correlated to ECA level ($p > 0.10$).

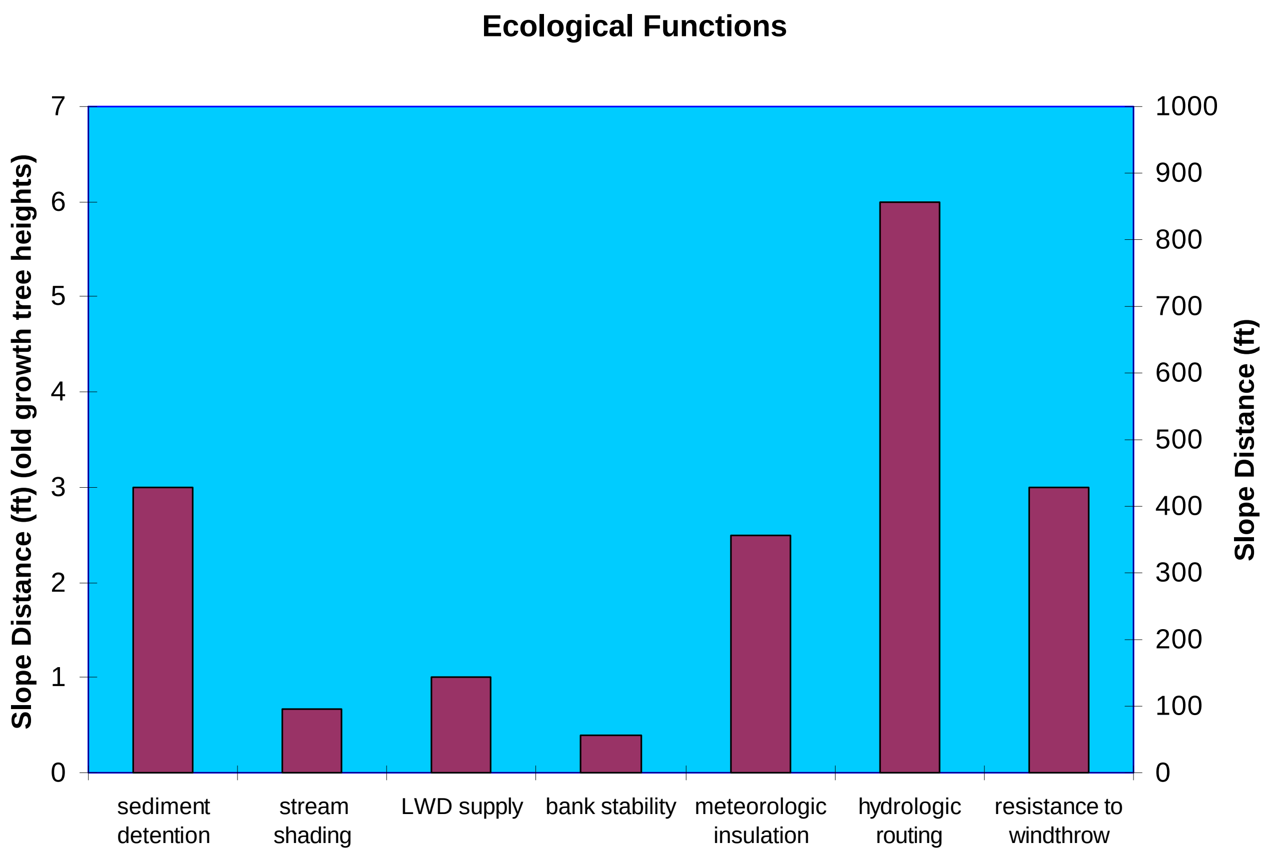

Figure 35. Estimated widths of protected areas, measured in slope distance from the edge of floodplain, needed to provide completely natural levels of ecological function over time with respect to some of the discrete ecological functions of riparian vegetation. Estimated slope distances in feet are based on the assumption that average old growth tree height is 150 feet. Widths of riparian reserves would have to extend to topographic divide to completely protect against increased sediment delivery during extreme events, alteration of hydrology, and increased susceptibility to windthrow.

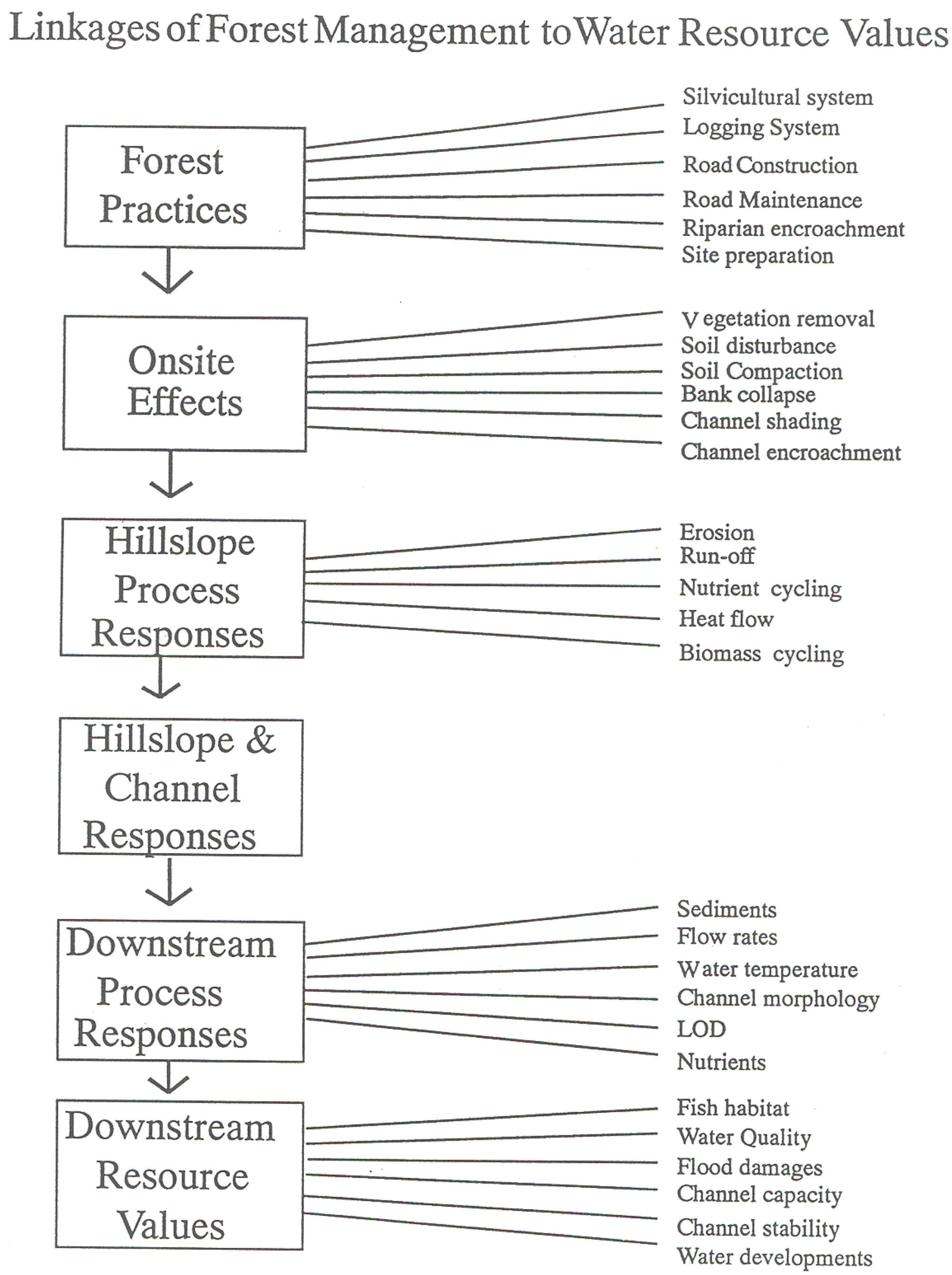

Figure 36. Generalized linkages between on-site logging activities and downstream habitat conditions, including factors affecting magnitude of on- and off-site impacts (After NCASI, 1992).

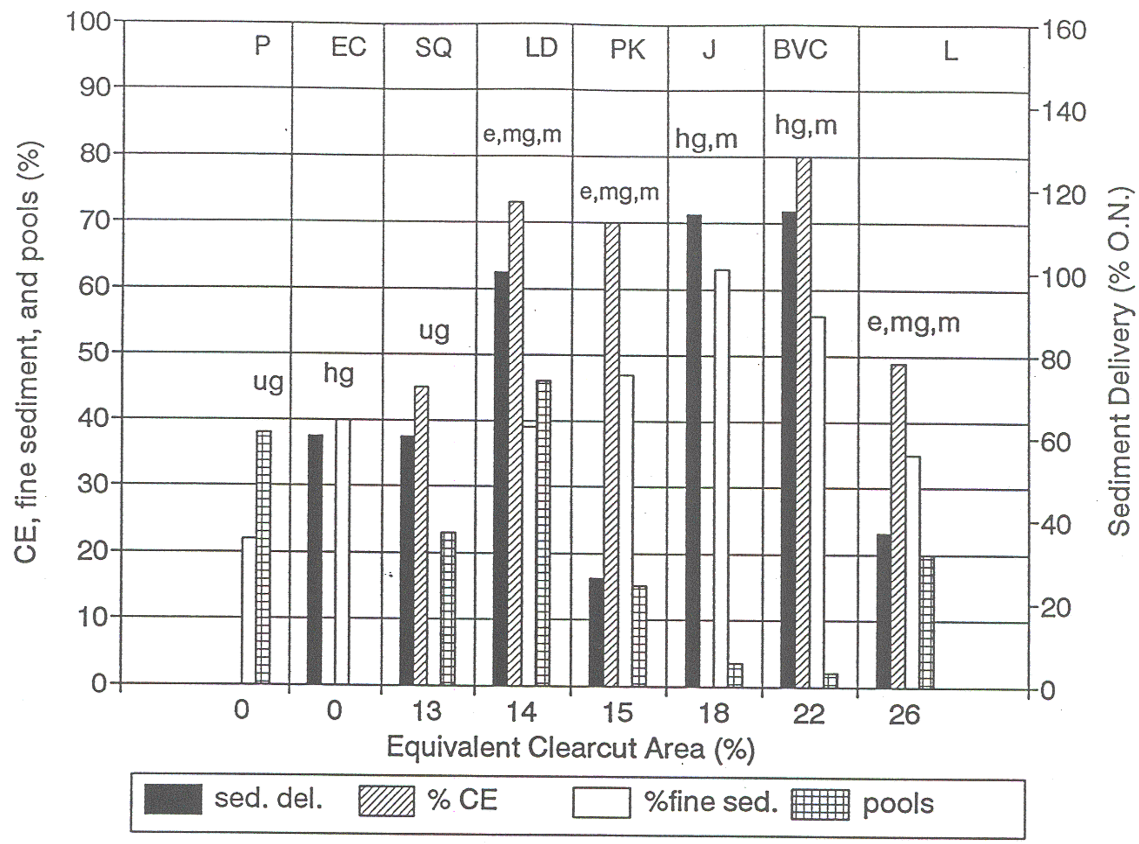

Figure 37. Habitat conditions, estimated sediment delivery, and Equivalent Clearcut Area (ECA) in watersheds in the Idaho batholith on the Clearwater (CNF) and Boise National Forests (BNF) of ECA. Abbreviations are as follows: ug=ungrazed; mg=moderately grazed; hg=heavily grazed; m=mined; e=enhancement efforts have been in place that may have affected the plotted variable; BVC=Bear Valley Creek (on the BNF); J=Johnson Creek (on the BNF) ; LD=Eldorado Creek (on the CNF); SQ=Squaw Creek (on the CNF); PK=Pete King Creek (on the CNF); P=Porter Creek (on the BNF); L=Lolo Creek (on the CNF); EC=Elk Creek (on the BNF). (Data from Espinosa and Lee, 1991; CNF, 1991a; CNF, 1992; CNF, 1993; BNF, 1993; NMFS; 1993; CNF, unpublished WATBAL runs). Cobble embeddedness (CE) data was unavailable for P, EC, and J. Pool data was unavailable for EC. Fine sediment data was unavailable for SQ. Although data generally indicate that habitat conditions deteriorate with increasing ECA (See Table 3 for R^2 values), the worst habitat conditions occur in watersheds that have been heavily grazed and mined, as well as logged. Substrate conditions are significantly degraded at ECA=0 in heavily grazed Elk Creek. PK, LD, and L do not have pool frequencies that fully reflect the legacy of land management because all three streams have had LWD added to form pools. It is not known how many pools have been formed by the enhancement efforts. Substrate conditions in PK also may not reflect the effects of land management because active sediment trapping and removal has occurred since 1989.

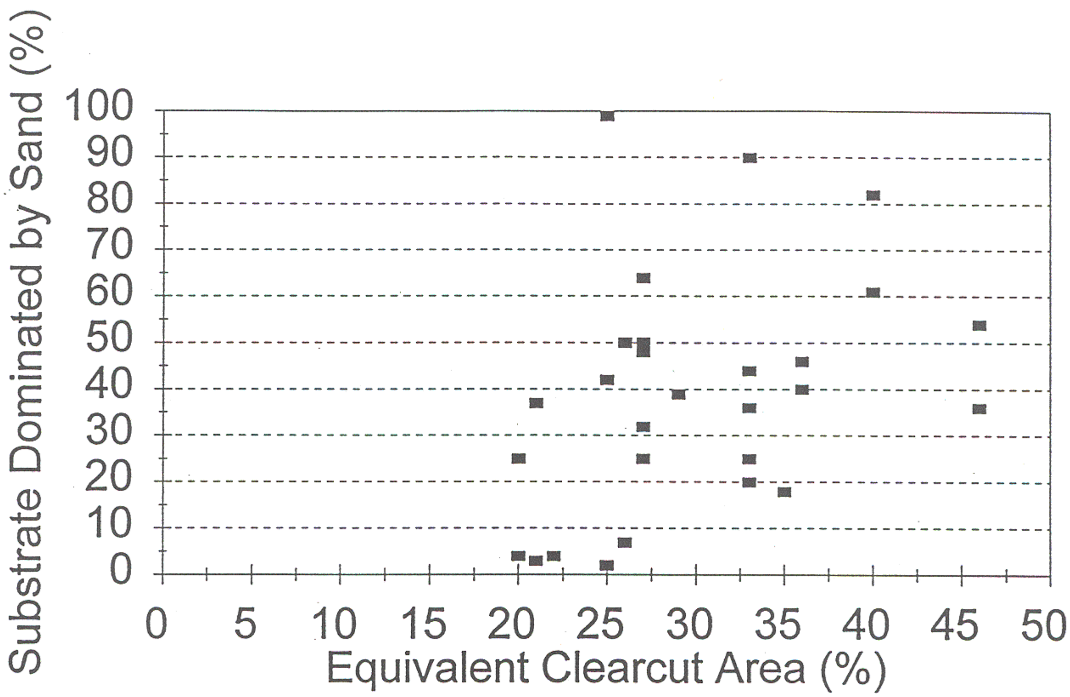

Figure 38. Percent channel substrate dominated by sand and Equivalent Clearcut Area subwatersheds tributary to the Grande Ronde River on the Wallowa-Whitman National Forest (WWNF) (WWNF, 1992). Substrate data are only for fish-bearing portions of the streams within the subwatersheds. All subwatersheds have been subjected to varying levels of grazing; some have been mined. Geology, elevation, area, topography, and vegetation vary among subwatersheds; no attempt was made to stratify data by subwatershed characteristics. Surface fine sediment levels greater than 20% have been considered to represent poor habitat conditions (McCammon, 1993). There are high levels of sand (>20%) in fish habitat at a wide variety of ECA levels. Sedimentation and fine sediment levels are a pandemic problem in salmon habitat in the Upper Grande Ronde River (Anderson et al., 1992). Causes of elevated sediment delivery include the combined effects of grazing, fire and flood, mining, road construction, and logging (Anderson et al., 1992). At these levels of fine sediment, sedimentation of cleaned gravels during the overwintering period is consistently high (Purser and Rhodes, in process). Although the correlation between percent substrate dominated by sand and ECA levels was weak (R^2=0.14), it is statistically significant ($p < 0.10$).

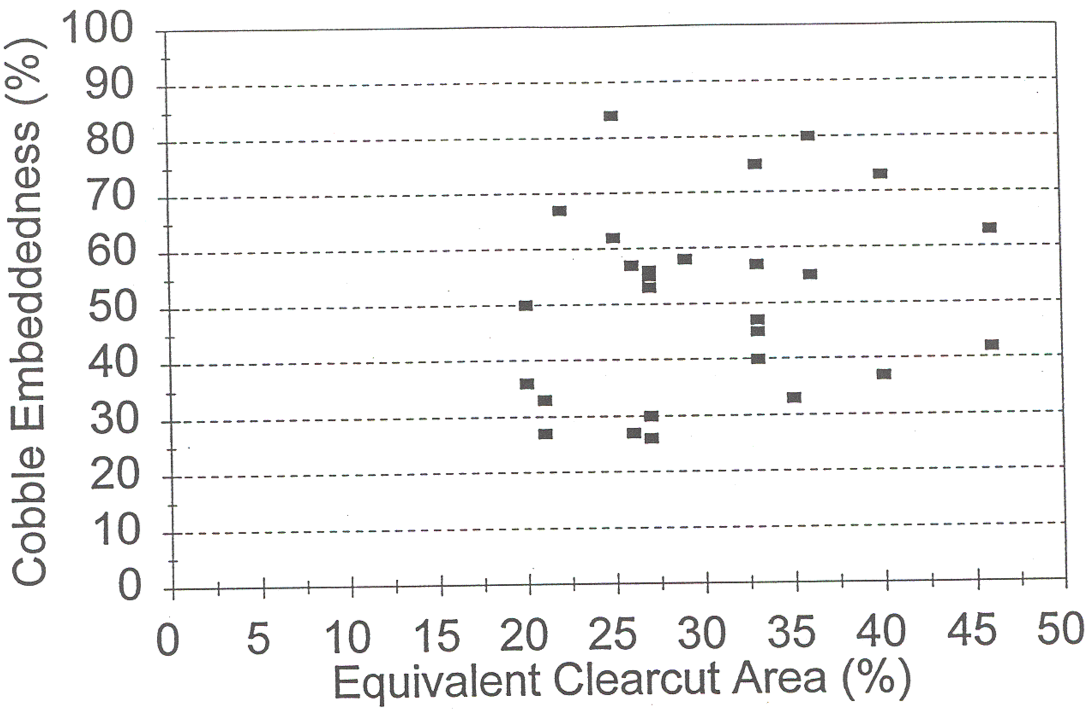

Figure 39. Cobble embeddedness (CE) in streams and estimated Equivalent Clearcut Area (ECA) in subwatersheds tributary to the Grande Ronde River on the Wallowa-Whitman National Forest (WWNF) (WWNF, 1992). Data are only for fish-bearing portions of the streams within the subwatersheds. All subwatersheds have been subjected to varying levels of grazing; some have been mined. Geology, elevation, area, topography, and vegetation vary among subwatersheds; no attempt was made to stratify data by subwatershed characteristics. CE levels greater than 35% have been considered to represent poor conditions (McCammon, 1993). At almost all levels of disturbance, CE is high (>30%). Data indicate that salmon rearing is severely impaired in many streams. CE levels in excess of 50% are expected to cause near total departure of fry (Chapman and McLeod, 1987); CE in excess of 70% causes significant alteration of the food web (Chapman and McLeod, 1987). Sedimentation and fine sediment levels are a pandemic problem in salmon habitat in the Upper Grande Ronde River (Anderson et al., 1992). Causes of elevated sediment delivery to the UGRR include grazing, fire and flood, mining, road construction, and logging (Anderson et al., 1992). CE is not significantly correlated to ECA level in these subwatersheds (R^2=0.04; $p > 0.10$).

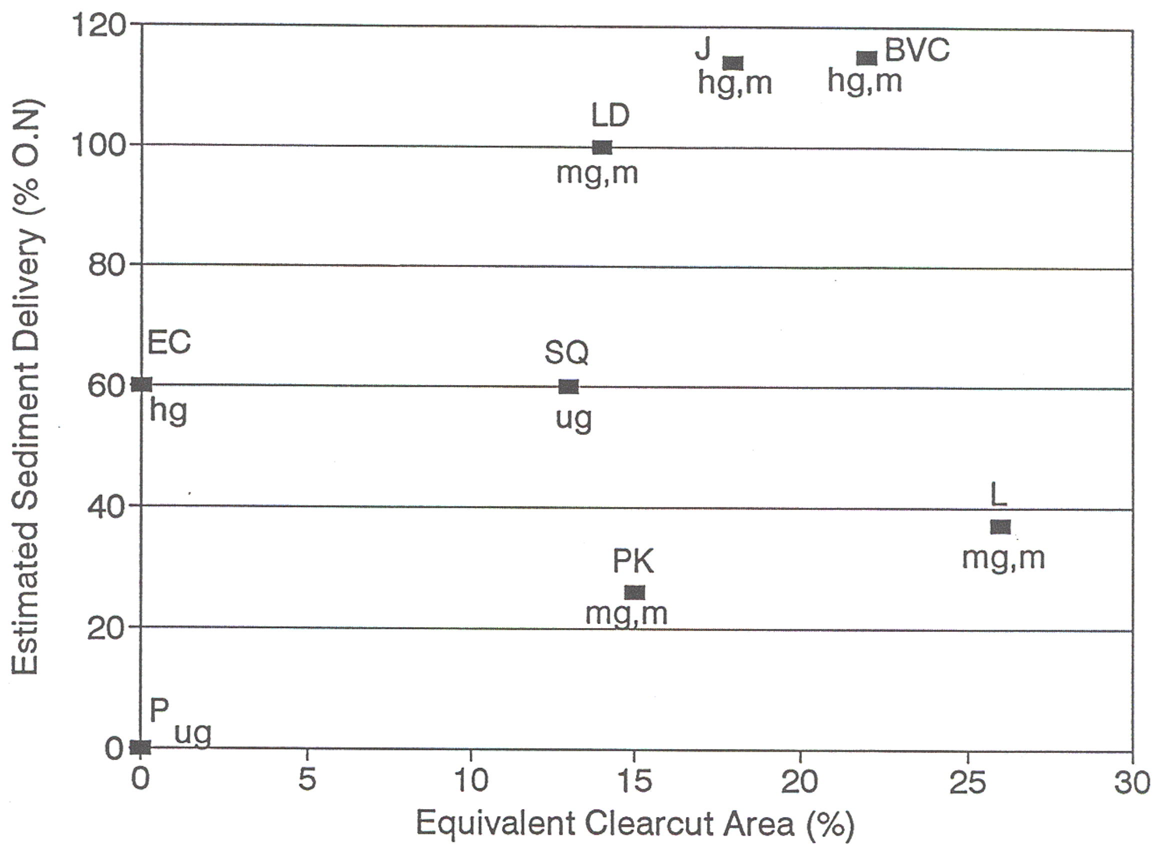

Figure 40. Estimated sediment delivery for watersheds in the Idaho batholith on the Clearwater (CNF) and Boise National Forests (BNF) at varying ECA levels. Where ECA levels were given as a range, the median within the range was used. Abbreviations are as follows: ug=ungrazed; mg=moderately grazed; hg=heavily grazed; m=mined; BVC=Bear Valley Creek (on the BNF); SQ=Squaw Creek (on the CNF); J=Johnson Creek (on the BNF); LD=Eldorado Creek (on the CNF); EC=Elk Creek (on the BNF); PK=Pete King Creek (on the CNF); P=Porter Creek (on the BNF); L=Lolo Creek (on the CNF); EC=Elk Creek (on the BNF). (Data from Espinosa and Lee, 1991; CNF, 1991a; BNF, 1993; NMFS; 1993). Estimated sediment delivery is not significantly correlated with ECA level (R^2=0.21; $p > 0.10$).

Table 1. Summary of effects of land use activities on key portions of the watershed system (in-stream, streambank, riparian, hillslope, watershed) and their linked effects on specific aspects of salmon habitat. Although land use activities can have numerous other effects on aquatic resources, primary attention was given to effects on habitat variables for which standards are proposed. Key: "▲ " = increase; "▼" = decrease; ➡" = "a function of;" CE = cobble embeddedness; LWD = large wood debris.

Land use activity	System component affected	Mechanism of effect	Habitat parameter affected
grazing	streambank	▲ bank erosion ➡ ▼ vegetation density and rate of regrowth; ▲ sediment delivery	▲ CE, ▲ surface fines, ▼ streambank stability
		▲ bank trampling and calving; ▼ bank stability	▼ bank overhang
		▲ summer water temperature, ▼ winter water temperature, ▲ daily and seasonal extremes, ▲ headward movement of critical thermal maxima	▲ thermal loading (summer) interception, ▲ heat loss in winter; ➡ ▲ channel width ➡ ▲ sediment load, ▲ bank trampling and ▼ bank stability;
		▼ wet meadow vegetation ➡ grazing, soil compaction;	▲ solar load interception
		▼ shrubs and trees, shift in vegetative community; ➡ ▼ shrub canopy, ▲ trampling and cropping of seedlings/saplings ➡ ▲ livestock grazing pressure	▼ stream shading
		▲ bank erosion, ▼ bank stability ➡ ▼ vegetation density and ▼ rate of regrowth; ▲ sediment delivery; ▲ physical disturbance of soil surface	▼ pool volume
		▲ solar radiation interception, ▲ longwave radiation emission from water surface ➡ riparian vegetation cropping and prevention of regrowth	▲ summer water temperature, ▼ winter water temperature, ▲ daily and seasonal extremes, ▲ headward movement of critical thermal maxima

Table 1 (Continued). Summary of effects of land use activities on aspects of salmon habitat.

Land use activity	System component affected	Mechanism of effect	Habitat parameter affected
grazing	floodplain	▼ groundwater baseflow ➡ ▲ channel incision, ▲ soil compaction, ▼ infiltration capacity, ▲ overland flow, ▼ water storage capacity, wetland damage	▲ maximum water temperature, ▲ water temperature extremes, ▼ flow velocity, ▲ intermittent reaches
		▼ wetland area ➡ ▲ streambed incision, ▲ baselevel lowering of wetland, ▲ drainage of wetland	▲ maximum water temperature, ▼ bank stability
		▲ upstream channel erosion, ▲ sediment load ➡ all streambank, riparian, and floodplain grazing effects	▲ CE and ▲ surface fines, ▼ pool volume and frequency
	hillslope	▲ peakflow ➡ ▲ vegetation removal by grazing, ▲ soil compaction	▲ sediment delivery, ▼ bank stability
logging	riparian	▼ stream surface shade, ▲ solar input, ▲ nighttime longwave radiation flux; ➡ ▼ vegetation cover	▲ summer water temperature, ▼ winter water temperature, ▲ daily and seasonal extremes, ▲ headward movement of critical thermal maxima
		▲ channel widening, ▲ streambank erosion; ➡ ▼ vegetation cover, ▲ peakflow, ▼ abundance of deep rooted vegetation, ▲ surface and mass erosion	▲ CE and ▲ surface fines, ▼ pool volume and frequency, ▼ streambank stability
		▲ channel and bank erosion, ▲ sediment delivery ➡ ▲ vegetation removal, ▼ LWD volume and recruitment to channel (▼ sediment storage)	

Table 1 (Continued). Summary of effects of land use activities on aspects of salmon habitat.

Land use activity	System component affected	Mechanism of effect	Habitat parameter affected
logging	riparian	▼ LWD volume and recruitment rate ➡ ▲ vegetation removal, ▼ size and number of trees	▼ pool volume and frequency, ▼ residual pool depth
		▲ pool sedimentation ➡ ▲ magnitude and frequency of channel erosion, ▼ LWD	
		▼ pool volume ➡ ▲ pool sedimentation and ▼ LWD; ▲ channel width ➡ ▲ discharge peaks, ▲ sediment delivery, ▼ streambank stability	▲ summer water temperature, ▼ winter water temperature, ▲ daily and seasonal extremes, ▲ headward movement of critical thermal maxima
		▲ vegetation cover, basal area, and deep rooting density, ▲ soil disturbance	▲ sediment delivery, ▲ turbidity
		▼ size and number of trees; ▼ LWD recruitment ➡ long recovery time of riparian trees	▼ LWD recruitment
	hillslope	▲ erosion and sediment delivery; ▲ mass failures ➡ ▼ rooting strength, ▼ tree cove	▲ CE and ▲ surface fines, ▼ pool volume and frequency
		▼ vegetation cover, ▼ groundcover, ▼ root strength, ▲ soil disturbance, ▲ mass and surface erosion, ▲ peakflow, ▲ overland flow	▲ sediment delivery, ▲ turbidity
		▲ peakflo ➡ ▲ vegetation removal, earlier snowmelt	▲ sediment delivery, ▼ bank stability
	watershed	▲ peak flows	▲ sediment delivery
		logging moratorium	▼ sediment delivery

Table 1 (Continued). Summary of effects of land use activities on aspects of salmon habitat.

Land use activity	System component affected	Mechanism of effect	Habitat parameter affected
roads	riparian	▼ stream surface shade, ▲ solar input, ▲ nighttime longwave radiation flux; ➡ ▼ vegetation cover	▲ summer water temperature, ▼ winter water temperature, ▲ daily and seasonal extremes, ▲ headward movement of critical thermal maxima
		▲ channel widening, ▲ streambank erosion; ➡ ▼ vegetation cover, ▲ peakflow, ▼ abundance of deep rooted vegetation, ▲ surface and mass erosion originating from road corridor	▲ CE and ▲ surface fines, ▼ pool volume and frequency, ▼ streambank stability
		▲ channel and bank erosion, ▲ sediment delivery ➡ ▲ vegetation removal, ▼ LWD volume and recruitment to channel (▼ sediment storage)	
		▼ LWD volume and recruitment rate ➡ ▲ vegetation removal, ▼ volume recruitment of LWD of large length and diamete	▼ pool volume and frequency, ▼ residual pool depth
		▲ pool sedimentation ➡ ▲ magnitude and frequency of channel erosion, ▼ LWD	
		▼ pool volume ➡ ▲ pool sedimentation and ▼ LWD; ▲ channel width ➡ ▲ discharge peaks, ▲ sediment delivery, ▼ streambank stability	▲ summer water temperature, ▼ winter water temperature, ▲ daily and seasonal extremes, ▲ headward movement of critical thermal maxima
		▲ vegetation cover, basal area, and deep rooting density, ▲ soil disturbance	▲ sediment delivery, ▲ turbidity
		▼ size and number of trees; ▼ LWD recruitment ➡ long recovery time of riparian trees	▼ LWD recruitment

Table 1 (Continued). Summary of effects of land use activities on aspects of salmon habitat.

Land use activity	System component affected	Mechanism of effect	Habitat parameter affected
roads	hillslope	▲ erosion and sediment delivery from road surfaces; ▲ mass failures originating from road system	▲ CE and ▲ surface fines, ▼ pool volume and frequency
		▼ vegetation cover, ▼ groundcover, ▼ root strength, ▲ soil disturbance, ▲ mass and surface erosion, ▲ peakflow, ▲ overland flow, ▲ sediment delivery, ▲ turbidity	
		▲ peakflow ➡ ▲ vegetation removal, earlier snowmelt, ▲ water routing from road drainage system	▲ sediment delivery, ▼ bank stability
	watershed	▲ road density, ▲ peak flows	▲ sediment delivery
		logging moratorium, ▼ road densit	▼ sediment delivery
mining	riparian	▲ erosion and transport of mine tailings, ▼ size and number of trees, ▼ LWD recruitment ➡ long recovery time of riparian trees	▲ CE and ▲ surface fines, ▼ pool volume and frequency
	hillslope	acid mine runoff, heavy metal pollution, sediment delivery	▼ water quality
	in-stream	gravel extraction	▲ CE and ▲ surface fines, ▼ pool volume and frequency, ▲ turbidity

Table 1 (Continued). Summary of effects of land use activities on aspects of salmon habitat.

Land use activity	System component affected	Mechanism of effect	Habitat parameter affected
agriculture	riparian	intense soil disturbance, ▼ vegetation cover, ▼ vegetation rooting density	▲ CE and ▲ surface fines, ▼ pool volume and frequency, ▲ sediment delivery
		feedlot runoff, runoff of agricultural chemicals (fertilizers, pesticides)	▼ water quality
	watershed	▼ stream summer baseflow ➡ irrigation withdrawal of diversion	▲ summer water temperature
		intense soil disturbance, ▼ vegetation density	▼ pool volume and frequency, ▲ sediment delivery

Table 2. Generalized, qualitative overview of channel response to changes in streamflow discharge and sediment yield. Abbreviations are as follows: Q=streamflow discharge; w=channel width; d=channel depth; R=width-to-depth ratio; L=meander wavelength; s=stream gradient; B=bedload sediment transport. Superscripted "+" and "-" indicate, respectively, an increase and decrease in the magnitude of the channel condition. After Schumm (1969).

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1) Increase in discharge only:

$Q^{+} \Rightarrow w^{+}\ d^{+}\ R^{+}\ L^{+}\ s^{-}$

2) Decrease in discharge only:

$Q^{+} \Rightarrow w^{-}\ d^{-}\ R^{-}\ L^{-}\ s^{+}$

3) Increase in bedload sediment transport only:

$B^{+} \Rightarrow w^{+}\ d^{-}\ R^{+}\ L^{+}\ s^{-}$

4) Decrease in bedload sediment transport only:

$B^{-} \Rightarrow w^{-}\ d^{+}\ R^{-}\ L^{-}\ s^{+}$

5) Increase in both discharge and bedload sediment transport:

Q^{+} and $B^{+} \Rightarrow w^{+}\ d^{+/-}\ R^{+}\ L^{+}\ s^{+/-}$

6) Decrease in both discharge and bedload sediment transport:

Q^{-} and $B^{-} \Rightarrow w^{-}\ d^{+/-}\ R^{-}\ L^{-}\ s^{+/-}$

7) Increase in discharge combined with a decrease in bedload sediment transport:

Q^{+} and $B^{-} \Rightarrow w^{+/-}\ d^{+}\ R^{-}\ L^{+/-}\ s^{-}$

8) Decrease in discharge combined with an increase bedload sediment transport:

Q^{-} and $B^{+} \Rightarrow w^{+/-}\ d^{-}\ R^{+}\ L^{+/-}\ s^{+}$

Table 3. Results of linear regressions between various pairs of variables from streams on the Clearwater and Boise National Forests in the Idaho batholith subject to various types of land use. In each column, the first number is the R^2 value followed by the number of sample pairs with level of statistical significance underneath in parentheses.

Independent Variables	Dependent Variables	% Fine Sediment by Depth	% Cobble Embeddedness	% Surface Fine Sediment	% Pools	Sediment Delivery (% Over Natural)	Bank Stability (%)
Equivalent Clearcut Area (%)		0.13, n =5 (*P* >> 0.10)	0.00, n = 5 (*P* >> 0.10)	0.42, n = 8 (*P* >> 0.10)	0.35, n = 7 (*P* >> 0.10)	0.21, n = 9 (*P* >> 0.10)	0.55, n = 9 (*P* >> 0.10)
Sediment Delivery (% Over Natural)		0.22, n = 5 (*P* >> 0.10)	0.34, n = 5 (*P* >> 0.10)	0.74, n = 9 (*P* >> 0.10)	0.16, n = 7 (*P* >> 0.10)	n/a	n/a
Bank Stability (%)		nd	nd	0.46, n = 9 (*P* >> 0.10)	nd	n/a	n/a
% Fine Sediment by Depth		n/a	nd	nd	0.64, n = 6 (*P* >> 0.10)	n/a	n/a

APPENDIX B

CASE HISTORY: THE FAILURE OF EXISTING PLANS TO PROTECT SALMON HABITAT ON THE CLEARWATER NATIONAL FOREST IN IDAHO

prepared by

F. Al Espinosa, Jr. (Consulting Fish Biologist)

INTRODUCTION

The large-scale loss of salmon habitat in the Snake River Subbasin and elsewhere in the Pacific Northwest is testimony to the failure of existing stewardship philosophies and their derivative management plans. Sedell and Everest (1990) and others (Pacific Rivers Council, 1992) have documented portions of the loss. In this section, the ***failure*** of past and existing management plans to protect salmon habitat on the Clearwater National Forest, Idaho, will be examined. Four natal tributaries of the Clearwater River Subbasin supporting salmon production have been selected for close examination of their case histories in terms of watershed and fish habitat impacts. These watersheds typify others in the Snake River Subbasin that have experienced similar histories of timber development. Moreover, the efficacy of past and existing management plans to protect and recover these watersheds will be investigated. The strategies, assumptions, prescriptions, and accountability features of these plans will be scrutinized.

BACKGROUND AND DESCRIPTION

Before 1973 and extending to 1987, management of the Clearwater Forest was guided by a generic document and policy called "Multiple Use Plan--Part I." This plan allocated the Forest landbase into general management zones such as the general forest zone, wild and scenic river zone, and riparian zone. A set of allowable management practices (guidelines) was described for each zone. These guidelines were extremely general in nature and subject to wide interpretation. For example, ***"protect the riparian management zone and dependent resources while harvesting timber"*** is the type of guidance that would be applied to the riparian zone. Specificity and quantification did not characterize the guidance. Other management policies, practices, guidance and direction available then were similar in nature and content. The Idaho State Forest Practices Act and implementing procedures offered no improvements because they consisted of excessively generalized guidance, escape clauses, and large loopholes.

The concept of ***"best management practices"*** (BMPs) was introduced as a catchall phrase representing the collection of all standard management practices of the day, which was hardly a guarantee of resource protection but were practices considered conventional, economical, reasonable, and acceptable to practitioners. In practice, resolution of resource conflicts was usually decided in favor of commodity interests at the expense of aquatic resources. Application of BMPs then became viewed as a justification for proceeding blindly with resource extraction when obviously insufficient attention would be given to assessing BMP effectiveness or monitoring trends of degradation in aquatic resources. Part I of the Multiple Use Plan called for more specific planning in Phase II of the process--unit planning (Clearwater National Forest, 1972). Large parcels of the Forest were stratified into management units incorporating parts of large watershed systems. Stratification was based mostly on timber availability, District administrative and roadless area boundaries rather than an ecosystem or total watershed approach. Extensive inventories of all resources were conducted and an attempt to quantify the resources was made. Resource information was then used to stratify and allocate the landbase into the "best" possible uses and management strategies. Resource outputs and a set of management practices were derived for each management unit within the plan. An attempt to resolve resource conflicts with unit policy statements was made. Unit plans were a definite improvement over Part I. They were based on resource information and did a much better job of allocating the landbase. However, they still lacked specificity, accountability, and failed to adequately deal with cumulative impacts.

Unit planning ended with the arrival of the National Forest Management Act of 1976 (NFMA). Only a few unit plans were completed and carried out. Under NFMA, the entire Forest was subjected to the planning process. More intensive inventories were conducted and an attempt to stratify the Forest landbase into somewhat homogeneous ecologic units was made. The Forest was stratified into management areas and allocated to the "best" use for that segment of the landbase. Goals, objectives, guidelines, standards, and prescriptions were developed for each management area. In addition, similar direction was developed for the Forest as a whole. Timber outputs were quantified for the management areas. Other resources were quantified from District and Forest summaries. Watershed and fish habitat quality standards were derived for most of the watersheds of the Forest. These standards were based on the quantified estimates of sediment yield and its effect on in-stream deposition of sediment--cobble embeddedness (Stowell et al., 1983). This was one of the few attempts in the Columbia River Basin to quantify fish habitat criteria and establish it as an ***"accountable"*** management standard. The Forest Plan was completed in 1987 and provides direction to the Forest's management efforts to this day. With seven years of implementation, several significant shortcomings have become obvious. Timber outputs were out of synchronization with watershed and fish habitat standards. Projected timber harvests and associated road construction levels were too high to maintain optimum water and fish habitat quality (Clearwater National Forest, Phase I Report, 1992 and Isaacson, 1994). The level of timber activity scheduled by the Forest Plan increased the extent and severity of degradation on the Forest (Clearwater National Forest, Phase I Report, 1992). Degraded watersheds that were projected to recover in the plan have not.

In 1992, the Forest documented the damage in its Phase I Report that listed some 90 developed drainages (71%) that were below Forest Plan standards for water quality. Isaacson (1994) has since corroborated this assessment in his review of Region One and Clearwater National Forest water quality data. These 90 watersheds have since been tentatively listed as ***water quality limited waterbodies*** by the State of Idaho in their response to the U.S. District Court's decision on Idaho's submitted list of streams for compliance under section 303 (d) of the Clean Water Act (Idaho Sportsmen's Coalition v. Browner, April 14, 1994).

There are many reasons why the Forest Plan failed to adequately protect the watersheds. Some of these are documented in the Forest's Phase I Report (1992). The primary basis of the failure resided in the modeling effort that linked FORPLAN (computer model) timber activities with WATBAL (watershed computer model, Patten, 1989) estimated impacts (sediment yield). The models were not integrated in that sediment modeling was conducted independently of FORPLAN and without a watershed perspective. Timber and associated sediment effects were estimated from broad (gross) management strata and were subjected to broadscale, inappropriate averaging (Clearwater National Forest Plan, 1987). ***Many planning assumptions of the modeling effort were simply invalid.*** The Plan assumed that degraded watersheds would display a significant recovery pattern within a short period (5 to 10 years). They did not. The pursuit of unattainable timber targets forced the Forest to reenter the damaged watersheds repeatedly. Data from modeling and monitoring efforts were ignored. Consistent funding for extensive watershed restoration did not materialize. Modeled trends in watershed sediment delivery and reduction of sediment delivery to levels approaching assumed ***"no impact thresholds"***--relative to dependent in-channel substrate conditions--did not agree with observed trends in substrate sediment conditions. The ***"no impact threshold"*** is a watershed-specific level of sediment delivery assumed by the Clearwater National Forest to potentially allow recovery in sediment conditions to occur. The observed increases in substrate fine sediments and cobble embeddedness or lack of recovery in these indices indicate that these modeling assumptions are faulty, that lag times for substrate response are very great, or that progress made in substrate recovery is easily reversed by periodic impacts. Existing roads (background sediment yield) were modeled to completely recover in six years. Monitoring and field observations proved this assumption erroneous. Management prescriptions, especially those for the riparian management area, were too generic and offered little protection. The original inventory of riparian areas was underestimated by a margin of some 86,780 acres (Phase I Report, 1992). The ***"light"*** scheduling of timber harvest in the riparian zones was consistently violated (Phase I Report, 1992). Uneven-age silvicultural treatment of riparian stands (another assumption) was seldom applied (Phase I Report, 1992). Water quality standards for developed watersheds were too obtuse and permissive. Criteria that described optimum fish habitat characteristics were not developed. Landbase allocations for fisheries resources were inadequate (Clearwater Forest Plan, 1987). Only 102,440 acres were allocated to the management of high value fisheries streams. A total of 503,567 acres was allocated to maximum timber development (Clearwater Forest Plan, 1987). Accountability for meeting water and fish habitat standards was not enforced. Although, the Forest Plan was a significant improvement over previous planning efforts, it still contained some ***"fatal"*** flaws concerning the management of fisheries resources and led to continued decline of anadromous and resident salmonids (Phase I Report, 1992).

The management history of the watersheds for ***Lolo, Eldorado, Pete King, and Squaw Creeks***, provide specific examples of failures of past and existing management plans to protect salmon habitat on the Clearwater National Forest. All of these watersheds have had similar histories (25+ years) of timber development and road construction. These watersheds drain portions of the Idaho Batholith, a large and sensitive geologic formation (granite) characterized by highly erosive terrain and rivers that are highly susceptible to sedimentation. Therefore, watershed and fish habitat conditions will be reviewed primarily from the perspectives of sediment delivery to fish habitat and resultant cobble embeddedness and substrate fine sediment levels. Other factors limiting salmonid production in these watersheds are sub-optimal temperatures, loss of riparian shade and potential large woody debris, low pool frequency and habitat diversity (Rich et al., 1992; Espinosa and Lee, 1991; and Phase I Report, Clearwater National Forest, 1992). Sediment and habitat data were provided by the Clearwater National Forest. Sediment yield data were derived from modeling efforts conducted on the Forest (Patten, 1989). Data on in-stream sediment conditions were obtained from field survey and monitoring programs (Phase I Report, Clearwater National Forest, 1992).

Lolo Creek is a large tributary of the mainstem Clearwater River, and Eldorado Creek is a principal tributary of Lolo Creek. Both watersheds are located on the Pierce Ranger District of the Clearwater National Forest. Pete King Creek is tributary to the lower Lochsa River, whereas, Squaw Creek is tributary to the upper Lochsa River near the Powell Ranger District (Figure B-1).

Lolo Creek

Lolo Creek, a seventh order stream, enters the mainstem of the Clearwater River from the north at river mile 54 and is 42 miles in length. The stream flows primarily in a south/southwesterly direction draining approximately 78.4 miles of existing and potential anadromous fish habitat. Of mainstem Lolo, 18 miles are within the National Forest boundary and drain a watershed of approximately 72,673 acres within the boundaries of the Forest. The remaining 24 miles traverse a mixed ownership pattern of private, state, Nez Perce Tribe, and the Bureau of Land Management interests. Our discussion and analysis will be confined to Clearwater National Forest ownership. The Lolo watershed has a range in elevation of 5238 ft at its headwater sources near Hemlock Butte to 1299 ft at its confluence with the mainstem Clearwater River. The stream displays a wide amplitude in its seasonal flow regime ranging from an average of 500 cfs during spring runoff to an average of 25 cfs during late summer flow (Espinosa and Lee, 1991).

Lolo Creek was once a significant producer of spring/summer chinook salmon in the Clearwater River Subbasin (Fulton, 1968; Chapman, 1981; and Espinosa, 1987). Chapman (1981) estimated that Lolo Creek was capable of producing 84,000 spring chinook smolts in its pristine condition. Today, it produces a mere fraction of this potential because of the combined effects of downstream mortality at hydroelectric facilities and degradation of habitat in the Columbia River ecosystem (Rich et al., 1992). In 1990, it was estimated by Rich et al.(1992) that Lolo Creek was seeded at 11% of its potential carrying capacity.

In concert with other integrated efforts to recover upriver stocks of salmon, massive hatchery supplementation has been conducted in the Lolo system. Despite the heavy stocking, escapement of adult chinook and densities of pre-smolt salmon remain at critically low levels in Lolo Creek (Espinosa and Lee, 1991). In the past few years, adult escapement has probably ranged from 50 to 75 fish (Murphy, pers. comm.).

The Lolo watershed has, over time, sustained manifold impacts from timber harvesting, road construction, mining, and grazing. In comparison to timber management, deleterious effects on fish habitat from placer mining (gold) and grazing remain at minor levels. The Lolo drainage has a lengthy history (30+ years) of timber management on the Forest. During this period, the allowable harvest has ranged from 15 to 30 million board feet. Hundreds of miles of logging roads have been built and thousands of acres have been harvested (primarily clearcut). Road construction and riparian harvesting have generated the most severe impacts on the aquatic habitats of the Lolo system. Excessive sedimentation, channel impingement, and elimination of large woody debris were the major impacts documented by Espinosa (1975) during his baseline habitat survey.

Management of the Lolo watershed (and the other three tributaries) will be reviewed from two periods--pre-1973 and post-1973. Stratification at 1973 is critical because that is the year that fish habitat monitoring was initiated on the Clearwater National Forest (Espinosa, 1975). The year of 1987 is also critical because that is when the Forest Plan of the Clearwater National Forest was approved for implementation.

Extensive timber development in the Lolo watershed started in 1957 as revealed by a sharp increase in sediment yield produced by logging road construction (Figure B-2). Sediment yield was estimated at 60% over natural in 1957 (WATBAL database, Clearwater National Forest). Before that date, sediment yield was below the Forest's ***estimated "impact threshold"*** of 35% for Lolo Creek. Figure B-2 displays a histogram of sediment yield before 1973. In concert with an accelerated road building program, sediment yield increased dramatically and sustained high levels through this period. Sediment yield ranged from a minimum of 60% over natural in 1957 to a maximum of 149% over natural during 1965-1969. The mean and median sediment yields were 120% and 122%, respectively. During this period, sediment yield did not exceed the ***"geomorphic threshold"*** (a level of excessive sediment that induces major channel changes such as braiding, deposition, and bank instability; it is interpreted as an index of watershed instability; Patten, 1989). However, sediment yield did not drop below the ***assumed "impact threshold"*** during this period. Road construction in Lolo's riparian zone significantly altered the stream's channel and induced instability (Espinosa, 1975). Streamside roads were significant and chronic sources of sediment to Lolo Creek and its tributaries (Espinosa, 1975).

Sediment yields gradually declined from 1969 to 1972. This was short lived as sediment production was accelerated to new highs from 1973 to 1976 by the continuation of logging and road building. Sediment levels exceeded 200% over natural in 1975 and 1976 (Figure B-2). Beginning in 1977 and extending to 1993, sediment production has declined (Figure B-2). From 1973 to 1993, the mean and median sediment yields were 102% and 64%, respectively. Only recently (1990-1993) have sediment yields approached the Clearwater's ***assumed "impact threshold"*** of 35%. In summary, Lolo Creek has experienced sustained, high sediment impacts from 1957 to 1983 (26 year period). During the last decade, sediment yields have approached 20 to 30% over natural sediment yields, a target considered essential on Lolo Creek before recovery in substrate conditions can commence (Clearwater Forest Plan, 1987).

In 1973 and 1974, fish habitat and population monitoring were initiated in the Lolo watershed (Espinosa, 1975). Before the 1970's, little habitat data were available although Murphy and Metsker (1962) observed and documented sediment problems in the Lolo watershed. In 1974, streambed coring of salmon spawning habitat was initiated in Lolo Creek and extended into 1983. From 1983 to the present day, other sediment and fish habitat variables have been monitored (Espinosa and Lee, 1991). By the late 1970's, feedback information from the monitoring program was starting to influence decisionmakers. The awareness of watershed and habitat degradation problems helped to initiate a moderation of timber and road construction impacts in the early 1980's. The Forest planning process and increased public involvement have further diminished the development program in the mid-1980's to the present day. Today any project involving extensive timber harvesting and road construction is swiftly challenged by the fisheries entities in the basin.

Substrate coring data collected from Lolo Creek from 1974 to 1983 are displayed in Figure B-3. These sediment levels in chinook spawning substrate far exceed conditions considered optimum (<20%) for chinook production (Bjornn and Reiser, 1991; Stowell et al., 1983). Because of the sustained, high sediment yields (1957-1973), levels of sediment (<6.4mm) in the spawning habitat ranged from 34% in 1975 to 43% in 1976 with a mean and median of 39% (95% confidence interval of +/-2%). There was little variation from year to year and no trend of recovery during this period. Sediment yields remained consistently high before and during the monitoring period. Despite the monitoring feedback and the obvious degraded habitat conditions, management continued to reenter the watershed with timber and roading projects. Although these projects were scaled down in relation to previous activities (less road building), they still generated impacts and stresses to the system. Given the impact history of the Lolo watershed, it may be extremely naive to anticipate recovery back to optimum conditions, especially under a scenario of reduced activity and continual reentry. Madej (1987) has calculated that it will take decades to centuries for excessive sediment to be flushed from some reaches of Redwood Creek, California. She further postulated that it may take centuries for complete mainstem recovery (i.e., removal of flood-deposited sediments; Madej, 1987).

After 1983, the substrate monitoring program was dropped because of timing and budgetary constraints. Fish habitat in Lolo Creek was then monitored periodically with a comprehensive transect methodology that measured channel structural elements plus selected sediment variables like cobble embeddedness and % surface fines (Espinosa et al., 1987; Espinosa and Lee, 1991; and Huntington, 1988 and 1993). In the early 1990's, a monitoring program measuring winter habitat quality and sedimentation was initiated. In 1983, the fish and riparian habitats of Lolo Creek were subjected to extensive rehabilitation efforts funded by the Columbia Basin Fish and Wildlife Program (Espinosa and Lee, 1991).

Espinosa and Lee (1991) have documented the details of the program. Rehabilitation primarily involved riparian restoration (planting of conifers) and in-stream efforts to increase habitat diversity with large woody debris and rock. Cattle were excluded from a critical reach via fencing. Evaluation of fish habitat and population responses to the rehabilitation has been an integral part of this program. Although increases in habitat diversity (pool quantity and quality) and spawning habitat were documented in Lolo Creek, cobble embeddedness remained essentially unchanged (54% in 1974 and 51% in 1988; Espinosa and Lee, 1991). However, statistically significant increases of steelhead and chinook parr in summer rearing habitat were observed for enhanced habitats over control habitats (Espinosa and Lee, 1991; Scully et al., 1990). Espinosa and Lee (1991) concluded that sediment conditions in Lolo's rearing habitat remain unchanged since the baseline survey.

The Clearwater National Forest reported in their Phase I Report (1992) that cobble embeddedness for Lolo was at 49%, a level that equates to roughly 60% habitat capability (the standard assigned to Lolo Creek was 80% habitat capability in terms of smolt production potential as effected by substrate sediment conditions). Huntington (pers. comm.) observed cobble embeddedness to be approximately 45% during his 1993 survey. Other habitat features such as pool quantity and quality remain well below optimum standards (Espinosa and Lee, 1991; Phase I Report, Clearwater National Forest). Monitoring of winter habitat quality from 1990 to 1993 revealed a decline of 54% in quality ("free winter particle") from baseline (Figure B-4; Espinosa, unpublished data). The quantity and quality of winter habitat in Lolo Creek are likely factors limiting production of anadromous salmonids in the system (Espinosa et al., 1987; Huntington, 1988). According to the Clearwater National Forest, the overall habitat quality of Lolo Creek was assessed at 63%, some 17% below their Forest Plan standard (Phase I Report, 1992).

Based on data for substrate conditions and sediment yield, and habitat capability analyses, it is concluded that essentially little or no recovery in habitat sediment conditions in Lolo Creek has taken place over a period of 19 years despite a substantial decline in sediment yield and extensive rehabilitation during this time.

<u>Eldorado Creek</u>

Eldorado Creek, a fifth order stream, is a principal tributary of Lolo Creek and enters the Lolo mainstem near the Forest Service boundary (Figure 1). The stream flows primarily in a south/south-westerly direction draining approximately 12 miles and 38.3 acres of anadromous fish habitat. The stream drains a watershed of approximately 29630 acres and is 26.5 miles in length. The Eldorado watershed has a range in elevation from 5399 ft near its headwater sources to 2850 ft near its confluence with Lolo. The entire watershed is within National Forest ownership. The stream is capable of providing 0.42 acres of suitable spawning habitat for chinook salmon. In 1984, passage barriers near its mouth were removed to ease upstream migration of chinook salmon (Espinosa and Lee, 1991). In 1985, the rearing habitat of Eldorado was rehabilitated with in-stream structures over an 8-mile reach. In the 1990s, the Clearwater National Forest constructed several sediment traps in small tributaries and initiated a sediment removal program (using a portable dredge). The habitat monitoring program in Eldorado has been restricted to periodic surveys and evaluation of habitat improvement projects. The Nez Perce Tribe has selected Eldorado as one of its production tributaries for its hatchery and supplementation programs. Since the early 1980s, thousands of pre-smolt and smolt chinook salmon have been stocked in Eldorado (Espinosa and Lee, 1991). During the summer of 1993, the first adult salmon was observed in the stream, presumably from these plants (Larson, pers. comm.).

Similar to the Lolo analysis, Eldorado's sediment history will be reviewed from the same time perspective--pre-1973 and post-1973. Extensive timber harvesting in the Eldorado watershed also started in 1957 as shown by a sharp increase in sediment yield produced by logging road construction (Figure B-5). Sediment yield was estimated at 80% over natural in 1957. Figure B-5 displays a histogram of sediment yield for the 20-year period from 1957 to 1993 (WATBAL database, Clearwater National Forest). Like Lolo, Eldorado's sediment production accelerated dramatically and sustained high levels through this period. Eldorado's sediment yields were much higher than Lolo's and exceeded the ***"geomorphic threshold" of 272% over natural*** four years during this period. Sediment yield ranged from a minimum of 80% in 1957 to a maximum of 341% in 1961 with a mean and median yield of 219% and 209%, respectively. According to the WATBAL model, when sediment exceeds the ***"geomorphic threshold"*** major channel changes and long term impacts can be expected. During this period, sediment yield did not drop below the ***Clearwater***

N.F.'s assumed "impact threshold" of 45% for this drainage. Similar to its parent stream, logging roads constructed in the riparian zones were significant and chronic sources of sediment to Eldorado Creek.

Starting in 1974, sediment production decreased slightly until 1977 when it again shot upward beyond its ***"geomorphic threshold"*** in 1978 and 1979 (Figure B-5). A recovery trend started in 1980 and lasted until 1985. Another pulse was observed from 1986 to 1989. Since 1990, sediment yield has gradually decreased but still remains above the ***"estimated thresholds"*** that must be achieved before recovery can commence in this stream (i.e., 20-30%). From 1973 to 1993, sediment yield ranged from 60% to 306% with a mean and median of 143% and 140% over natural, respectively. Starting in the late 1970s, feedback based on surveys and the forest planning process helped create some awareness and sensitivity to the worsening condition of fish habitat. During the 1990s, this awareness and the public's involvement have helped to moderate the development program in Eldorado. ***However, the Clearwater National Forest continues to plan additional timber sales and roading in this below standard watershed that could further contribute to its degraded condition and delay the onset of a recovery trend.***

The fish habitat of Eldorado has been surveyed periodically as part of the Clearwater National Forest's ongoing fisheries program. In the Forest Plan database, Eldorado's level of cobble embeddedness is listed at an average of 37%. The data were collected in the 1970s. In the mid-1980s, Eldorado was resurveyed and cobble embeddedness was measured at a range of 50%-60% in salmon habitat. For the Phase I Report (1992), cobble embeddedness was assessed at an average of 73% for all surveyed reaches. Some critical reaches are in slightly better shape with a range of 45% to 60% cobble embeddedness (Huntington, 1992). In summary, surveys from the 1970's to 1990's have shown an increasing trend in CE, which likely commenced with the large peaks in sediment yield that occurred as the watershed was developed and continued with further development of the watershed.

Other habitat problems have been identified in Eldorado Creek by Vogelsang et al.,(1985). Some of these problems were: low pool frequency, poor pool quality, low levels of large woody debris, poor winter habitat, lack of substrate diversity, low levels of in-stream cover, and high water temperatures during critical rearing periods. Most of these habitat conditions were associated with excessive harvesting of riparian timber and poor road construction.

Obviously, the Eldorado system has been subjected to a long history of severe sediment and riparian impacts. Today its salmon habitat remains in a significantly degraded condition. Its condition is well below its Forest Plan standard of 80%+ habitat capability based on cobble embeddedness alone. A cobble embeddedness <35% is required to achieve habitat capability of 80% for a stream having a C-channel type (Espinosa, 1992). Eldorado Creek has been listed (tentatively) as a ***water quality limited waterbody*** by the State of Idaho and the Environmental Protection Agency (Martin, pers. comm.; State of Idaho's draft (303) (d) list submitted to EPA, May 13, 1994). Nevertheless, additional timber and roading projects are being planned for the Eldorado watershed. Despite extensive watershed and in-stream rehabilitation plus a relative decrease in sediment yield, there has been no recovery of watershed and habitat conditions.

Pete King Creek

Pete King Creek is a fourth order tributary of the lower Lochsa River near its confluence with the Selway River (Figure 1). The Lochsa River was once a significant producer of steelhead trout and spring chinook salmon (Espinosa and Lee, 1991). Pete King drains a watershed of some 17,526 acres and has a channel length of 12.5 miles. Its headwaters are located near Woodrat Mountain. The stream has a range in elevation of 1476 ft to 5218 ft. Pete King is entirely within National Forest ownership except for a small patented mining claim near its mouth. The stream was surveyed by Huntington (1992) who estimated that it could provide some 17.8 acres of rearing habitat for anadromous fish. Huntington (1992) observed only 0.0077 acres of spawning habitat for chinook salmon, most of which he classified as poor. Because of its steep gradient, chinook spawning would likely be confined to the lower 4.5 miles of mainstem Pete King. Some summer rearing of juvenile chinook does occur in the lower reaches of Pete King. However, because of excessive summer rearing temperatures (>68°F), Pete King is, at best, marginal salmon habitat.

Pete King has a long history of logging, mining, and grazing impacts. For the past 20 years, mining and grazing have had a very minor influence on the watershed. Since the mid-1950s, logging and associated road construction have been the predominant activities in the watershed. During its development history, the headwaters and riparian areas of Pete King have been subjected to heavy impacts. Pete King was also subjected to large, intense fires in the early 1900s. Starting in the 1970s, the watershed and habitats of Pete King have undergone extensive rehabilitation. In the mid-1980s, extensive in-stream restoration of spawning and rearing habitats was initiated under the auspices of the Columbia Basin Fish and Wildlife Program (Espinosa and Lee, 1991). Several in-stream sediment traps were also constructed in mainstem Pete King and its principal tributaries during this period. Sediment is removed annually from these traps and habitat is monitored for recovery in downstream reaches. Habitat and populations have been monitored in Pete King since the late 1970s. Recently, Pete King Creek has been selected as a research stream for chinook salmon supplementation (Bowles and Leitzinger, 1991).

Logging and associated road construction in Pete King Creek were initiated in the mid-1950s and then accelerated in the late 1950s to the mid-1960s (Figure B-6). Sediment yield was estimated at 65% in 1955 and then reached its apogee in 1963 at 347% over natural (Figure B-6). From the period of 1961 to 1974, sediment yields were sustained at levels exceeding the ***"geomorphic threshold"*** of 174% over natural. A histogram for the two periods of pre-and post-1973 displays drastically different scenarios (Figure B-6). From 1955 to 1972, sediment yield averaged 197% with a median yield of 220% over natural. From 1973 to 1993, the mean and median sediment yields were 77% and 54% over natural, respectively.

Since 1973, sediment yield in Pete King has declined significantly as road construction dropped-off sharply although it still remains elevated. Timber harvesting and roading did not completely stop. To this day timber projects are being carried out and planned for the watershed. In the 1990s, sediment yield has approached levels of <35% over natural, a level considered necessary for recovery to be initiated. Without monitoring and feedback to the management system, it is likely that sediment yields would have remained at levels of 60% to 100% over natural during the decade of 1983 to 1993. The Pete King Creek monitoring program was conducted at a frequency and intensity great enough to provide feedback to management on impacts to fish habitat so irrefutable that the sustained impacts of an aggressive timber program could no longer be overlooked.

Habitat problems in Pete King have been documented by many investigators (Murphy and Metsker, 1962; Talbert and Espinosa, 1986; Espinosa and Lee, 1991; and Huntington, 1992). Sediment, temperature, and the lack of structural diversity are just a few of the more salient problems. Concerning chinook salmon, sediment in winter habitat and temperature are probably the most critical limiting factors in the system (Espinosa and Lee, 1991 and Huntington, 1992). In the Forest Plan database, Pete King is listed with an average level of cobble embeddedness of 47%. In the early to mid-1980s, observed cobble embeddedness ranged from 50% to 60% (Talbert and Espinosa, 1986). Huntington (1992) recorded an overall average cobble embeddedness of 53% that included the West Fork. The mainstem of Pete King below the confluence of its forks displayed a mean cobble embeddedness of 36% (Huntington, 1992). A considerable amount of sediment still remains in the tributaries to Pete King. The West Fork showed a range in cobble embeddedness of 45% to 87% among its reaches. Three principal tributaries (Nut, Placer, and Walde) displayed a range of 42% to 45% in cobble embeddedness (Huntington, 1992). Old logging roads continue to fail and deliver sediment to the tributaries and mainstem of Pete King Creek (Stotts, pers. comm.). Although there is some evidence that summer and winter rearing habitats in the mainstem are showing some signs of recovery (Huntington, 1992 and Clearwater National Forest Monitoring Report, 1992), regression analysis indicates that there has been no statistically significant recovery trend from 1985 to 1992 despite sediment trapping (See Figure 7 in Appendix A).

Coring of spawning substrate was initiated in the mid-1980s. Figure B-7 displays a histogram of percentage fines (<6.4mm) in spawning substrate of mainstem Pete King during the period from 1985 to 1993. During this period, fines ranged from 30% to 47%, with mean and median levels of 37% and 36%, respectively. These levels are well above those considered optimum for salmonid survival (i.e., <20% fines <6.4 mm) (Stowell et al., 1983; Tappel and Bjornn, 1983). Despite a significant decline in sediment yield from 1973 to 1993, in-stream fines have remained high. The histogram displays two short periods of recovery interrupted by a three-year period of increase from 1989-1991. In-stream fines have declined in 1992 and 1993 (B. Stotts, pers. comm.). The short-term decrease in fines may be attributed to sediment trapping and removal which commenced in 1986 A large sediment trap is located about 0.5 mile above the substrate transects for mainstem Pete King Creek. This brief respite may be short-lived as new projects are being planned for the watershed and old logging roads continue to deliver sediment to the tributaries (B. Stotts, pers. comm.).

Despite a decline in sediment yield and extensive rehabilitation, habitat data indicate that recovery of degraded substrate conditions in fish habitat has not taken place in Pete King Creek (Figure B-7; See also Figure 7 in Appendix A).

<u>Squaw Creek</u>

Squaw Creek is a fourth order tributary of the Upper Lochsa River (Figure 1). This stream was once a significant producer of steelhead trout and spring chinook salmon (Espinosa and Lee, 1991). It drains a watershed of some 17,267 acres and has a mainstem channel length of 4 miles. Squaw Creek is estimated to provide 18.7 acres of rearing habitat and 0.73 acres of spawning habitat for anadromous fish (Espinosa and Lee, 1991). Although Squaw Creek exhibits the same general pattern of historical timber development, there are some differences between it and the three previously discussed streams. Extensive timber harvesting and roading were initiated in the mid-1950s. In the mid-1980s, extensive in-stream and riparian rehabilitation were conducted in the mainstem and the East Fork. Squaw Creek was also selected as a research test stream for chinook salmon supplementation and is currently being evaluated by the Nez Perce Tribe (Bowles and Leitzinger, 1991).

Road construction associated with logging started in 1956 when sediment yield was estimated to exceed the ***assumed "impact threshold"*** of 45% over natural and attained a level of 62% over natural. From 1956 to 1963, sediment yield stayed in the range of 60% to 100% over natural (Figure B-8). In 1964, it increased to 125% and reached levels of 293% and 372% over natural in 1969 and 1970, respectively (Figure B-8). During the pre-1973 period, sediment yields exceeded the ***"geomorphic threshold***" of 207% over natural for three years and showed a mean and median yield of 148% and 123% over natural, respectively. Starting in 1972, sediment yields declined slightly until 1979 when another increasing trend was initiated (Figure B-8). From 1982 to 1985, sediment yields again decreased slightly. From 1986 to 1988, an increasing trend was again observed. Since then, sediment yields have declined slightly approaching ***"assumed recovery thresholds"*** (<30% over natural) in the 1990s at which point continued improvement in substrate fine sediment can be expected to commence. From 1973 to 1993, the mean and median sediment yields for Squaw Creek were 86% and 87% over natural, respectively. Sediment did not exceed the ***"geomorphic threshold"*** during this period. Today additional timber sales are being planned for the Squaw Creek drainage.

In the Forest Plan database, the mean level of cobble embeddedness for Squaw Creek is listed at 50% based on surveys conducted in the 1970's. Kramer et al. (1985) documented a range of 30% to 40% cobble embeddedness before in-stream rehabilitation. The latest habitat assessment by the Clearwater National Forest listed cobble embeddedness at 45% for mainstem Squaw above Doe Creek (Phase I Report, 1992). Other habitat problems in Squaw are associated with poor riparian management (Kramer et al., 1985 and Espinosa and Lee, 1991). Excessive harvesting and riparian road construction have altered and simplified a great deal of the rearing habitat in Squaw and its tributaries. In many areas, the damage is permanent. In-stream restoration associated with the Columbia Basin Fish and Wildlife Program has attempted to alleviate some of these problems by introducing large woody debris and rock. Some riparian habitat was recovered when the parallel riparian roads were rehabilitated and reconstructed. Espinosa and Lee (1991) have documented the habitat and population responses to this effort. Nevertheless, the overall habitat quality of Squaw Creek remains at 63% on the basis solely of cobble embeddedness, some 17% below its Forest Plan standard (Phase I Report, 1992). Squaw Creek has not recovered from its history of development.

Summary Analysis

The history of watershed development and resultant changes in substrate sediment and habitat condition of four salmon watersheds on the Clearwater National Forest have been examined in detail. The analysis has demonstrated very similar patterns of timber development and associated impacts to salmon habitat. Other watersheds on the Clearwater National Forest have a similar history with comparable results including drainages such as Yoosa, Deadman, Papoose, Crooked Fork, and Brushy Fork (Phase I Report, 1992). This is certainly not unique to the Clearwater National Forest.

Without much difficulty, one could document analogous situations elsewhere in the Snake River Subbasin. Available data and analyses consistently indicate that the vast majority of watersheds managed for ***"multiple uses"*** exhibit degraded conditions in their fish habitats (Sedell and Everest, 1990; Platts and Chapman, 1992; and Isaacson, 1994).

Why have past and existing management plans failed to protect salmon habitat? In the early days (1950s to mid-1970s), the management strategy that was applied can be best characterized as ***"management by implied generalization."*** Essentially, the watersheds and habitat will take care of themselves despite practices and impacts.

The axiomatic management assumptions derived from this management perspective are the concepts of ***"free lunch"*** and ***"immaculate recovery."*** Translated, these amount to assumptions that whatever the current status of the watershed or fish habitat, there is a certain amount more impact that can be exacted without serious biological or physical consequences to system integrity. At least the impacts cannot be detected over the background of existing impacts; in addition, despite how serious the historic impacts have been, management today is conducted according to much refined practices that will result in a slow glide path to full recovery, as long as we continue to implement BMPs long enough. Unfortunately, the recovery scenario as practiced does not aim at recovery as a primary target. Rather, recovery is an anticipated spin-off from further development. For example, by harvesting trees in a floodplain, rehabilitation funds for use in the riparian zone can be generated. Or, rather than building a new primitive unsurfaced road, gravel is added to the road surface to lower sediment delivery in a watershed where cumulative sediment delivery is already above impact thresholds. BMPs then are merely means to reduce the level of impact given a decision to proceed with development.

Past management policies and practices led to extensive development and severe impacts that managers are still trying to deal with today. There was no ***"free lunch" or "immaculate recovery.***" An integral part of that prevailing management effort that has persisted into today's thinking is the idea that BMPs adequately protect aquatic resources. Stanford and Ward (1992) have labeled the ***BMP paradigm*** as a prime example of the ***"illusion of technique"*** process that is in vogue today (R. Behnke, Colorado State University, as cited in Stanford and Ward, 1992). The authors describe the process as a mere formalization and synthesis of ***"best professional judgment"*** with no ecological rationale that is empirically based. Specific sites are visited and subjective judgments are rendered on whether impacts to the resources have occurred. Additionally, inferences as to the significance of the impacts (or lack of impacts) are usually made (Stanford and Ward, 1992). The assessments are strictly subjective, qualitative, and cryptic.

A great deal of the failure to protect salmon habitat can be attributed to this philosophy and ***illusion***. It could be more appropriately named ***"least management practices."*** BMPs are subject to a wide spectrum of interpretation--frequently by disciplines not qualified to apply measures to protect salmon habitat or that have other resource objectives in mind. Therefore, the least effective practices are frequently applied. BMPs are contingent upon economic considerations and are habitually diluted or dropped because they are not ***economically feasible.*** BMPs do not deal with cumulative effects and the recovery of impacted watersheds. In fact, they promote cumulative effects and do not allow recovery because there are no watershed or fish habitat standards (criteria) to regulate or stop the application of practices. As long as BMPs are applied, habitat conditions are assumed to be fine regardless of existing watershed conditions and regardless how much land is subjected to impacts provided that BMPs are employed. Subjective assessments are too easily influenced by managers looking for facile answers to complex problems (***the free lunch syndrome).*** Mechanistically, the concept functions like a perpetual motion machine. BMPs cannot protect a watershed from excessive development. This philosophy has unequivocally failed to provide adequate protection for salmon habitat.

There are other examples of reliance on the ***"illusion of technique***" process that have led to widespread degradation of watersheds and fish habitats. The FORPLAN debacle is a good example. This simplistic computer model was strictly a timber-producing prototype built on a mountain of invalid assumptions. Because it was a computer model, it was given a ***sanctified status*** by forest planners that was inviolate. This deified model drove the process of setting timber harvest expectations. The FORPLAN model was not grounded in reality nor had it been validated and calibrated (tested) against real data. Timber volume projections used in the model were not ***"ground-truthed"*** against actual production and standing volume data by landscape strata (Clearwater National Forest Plan, 1987 and Phase I Report, 1992). Because of its inherent weaknesses and constraints, only gross resource stratifications (land capability units) were possible (Clearwater National Forest Plan, 1987). A watershed or ecosystem perspective was not applied to the process. Therefore, there was no integration of resource capabilities, values, or impacts. An attempt was made to link external watershed modeling efforts to the FORPLAN process; however, it was unverified and still subject to gross stratifications (Clearwater National Forest Plan, 1987). The result was faulty conclusions concerning resource outputs, scheduling of timber sales, impacts, recovery scenarios, and the attainment of water quality standards. Totally unrealistic timber volumes were projected for scheduling and harvest (Clearwater National Forest Plan, 1987). It soon became apparent that the targets were unattainable and that water quality standards for the developed watersheds were not being met (Phase I Report, 1992). The plan was litigated by the Sierra Club and the Wilderness Society in February 1993 and a negotiated settlement was reached in October 1993. The prime issues in the litigation were unrealistic timber harvest projections, uncertainties in the water quality modeling, and incomplete water quality standards (Clearwater National Forest, letter to the public, dated October 8, 1993). The Clearwater National Forest agreed to set a timber volume ceiling of 80 million board feet (MMBF) and to not cause further impacts in its ***"below standard"*** watersheds (***op.cit.***, ***1993)***.

Other significant manifestations of the illusion of rigorous analysis, promoted by FORPLAN modeling, involved projections of watershed recovery rates under extant levels of watershed impacts, mitigation measures, impact thresholds associated with each watershed, and availability of rehabilitation funding that could be counted on to ameliorate ongoing habitat damage. There was little or no data to support any of these contentions. Recovery of heavily degraded watersheds was a pivotal assumption in the Clearwater National Forest plan. The recovery assumption was based on the premise of receiving funding to abate sediment sources (i.e., road obliteration) and mitigate impacts. The funding did not materialize in a timely, consistent manner or adequate amount. At the same time, the assumption that these ***"below standard"*** watersheds could still be re-entered with ***moderated*** logging activities without interrupting recovery was accepted by management.

The Clearwater Forest believed that modeling and mitigating expected sediment yield to levels below ***"impact thresholds"*** would make it possible. Again, simple answers to complex problems that were not verified by data (e.g., measurement of sediment delivery, in-channel substrate sediment condition) were accepted. The ability of site-specific mitigation measures to reduce erosion and sediment delivery had not been adequately quantified. Modeling of surface sediment erosion on existing road networks was based on assumptions of total extinction of accelerated sedimentation in six years. This did not happen. Extant roads continue to experience accelerated surface erosion. ***"Impact thresholds"*** were not calibrated or validated with empirical evidence. In dealing with severely degraded watersheds and habitats, it is likely that ***zero*** sediment delivery over natural is the appropriate threshold for recovery (Heede, 1980). With degraded watersheds, the notion that moderated logging would allow recovery simply did not work (Phase I Report, 1992). Watersheds did not recover or they were further degraded (***op.cit.***, 1992).

The inability of the Forest to promote recovery of damaged watersheds while attempting to achieve timber harvest schedules known to be unjustified considering extant fish habitat conditions was further exacerbated by an ***"escape clause"*** standard. This particular standard allowed a one-time variance in the "***recovery***" pattern in drainages that were below their respective standards. This allowed many timber sales to survive that otherwise would have been considered a violation of Forest Plan standards and proceed toward implementation. "***Recovery***" was predicted from modeling efforts and not real data. In addition, modeled recovery in the distant future was traded for near-term continued improvement in habitat quality. Monitoring of in-stream sediment conditions soon established that ***modeled recovery scenarios*** were not accurate (Phase I Report, 1992). The escape standard was used for several years before it was declared illegal by the Regional Office and dropped by the Forest.

Although no documented or consistent standard was established as a replacement in the Forest Plan, management has used and continues to use several variations on the theme of recovery to make their decisions. The latest interpretation is ***"no measurable increase in sediment production in drainages currently NOT meeting Forest Plan Standards"*** (Clearwater National Forest, letter to the public dated October 8, 1993). In practice, this effort has become a procedure to rationalize timber sales in watersheds where conditions violate Forest Plan standards. Under such a standard continued impacts are allowable provided that they cannot be statistically detected against the existing background of degraded habitat. Road construction and timber harvesting have been moderated; however, impacts have continued to be placed on the drainages. Many projects have been contested and appealed by the public. Most proposals have been significantly modified, delayed, or dropped (Clearwater National Forest monitoring reports, 1990-1992). Without public intervention, it is very likely that the damaged watersheds would have been subject to still greater impacts.

It is only recently that sediment yields in some drainages have approached levels where the initiation of recovery might be expected. The estimated sediment levels have decreased to 20 to 30 % over natural category. However, these sediment delivery levels may still be too high to allow recovery. If more logging occurs within these watersheds, it can be expected that sediment delivery will increase and further degrade habitat and delay recovery. ***It is our opinion that sediment yields must remain well below the 20% over natural level until recovery of watersheds and fish habitat***

is documented by "hard data."

Certainly, the advent of the Forest Plan with its quantitative standards for watersheds and fish habitat on the Clearwater National Forest has prevented the continuation of severe resource damage that was observed in the 1950s through the 1970s. Nevertheless, there are major flaws, omissions, and loopholes in this management philosophy and strategy that have prevented salmon watersheds and fish habitat from significant recovery. ***In the following list, articulates the "management corrections" that are necessary to adequately protect salmon watersheds***.

Management Corrections

- Apply the ESA screening process recommended by this document throughout the Snake River Basin and extend it to the Clearwater, Umatilla, and John Day River Subbasins in order to protect refugia for colonists to and from critical habitat for spring/summer chinook.
- Establish Forest Plan standards for salmon watersheds based on our screening criteria for watersheds and fish habitat.
- Maintain roadless areas in salmon watersheds roadless until there is documentation that 90% of managed salmon watersheds have recovered to optimum levels ***(the roadless reserve concept)***.
- Take all of the riparian areas out of the allocated timber and grazing landbase. In salmon watersheds, this would mean that fish resources would be the primary, dominant and dependent resources for that management area. No activity would be permitted in this management area that did not enhance or maintain the fish resources. Timber sales and grazing would not be scheduled in this management area as part of the programmed harvest and grazing regimes.
- Projects programmed for ***degraded or "below standard" watersheds*** should not proceed until they can meet the screening criteria for watersheds and habitat.
- Continue to monitor watersheds, fish habitat, and populations to provide feedback on recovery progress.
- Eliminate roads in landscapes prone to severe surface and mass erosion.
- Conduct comprehensive watershed and fish habitat analyses plus inventories of rehabilitation needs with interdisciplinary teams. Establish a basin-wide priority list for watershed and fish habitat restoration projects. Subject the "***need***" for fish habitat restoration to a limiting factor analysis. ***Prior to any active restoration activities, remove or reduce anthropogenic perturbations to levels that allow recovery. Emphasize watershed (including riparian) restoration activities as a first priority before considering active in-channel or off-channel treatments.***
- Validate BMPs, mitigation, and restoration measures with hard data and science; subject these management techniques to peer review; and develop specific BMPs for salmon watersheds (including private and state land)that feature dramatic reduction of risk to the stocks, rapid habitat recovery, unimpeded recovery, and no grace periods in which one last sediment delivery spike is allowed before recovery proceeds.
- In conducting analyses of cumulative effects in large watersheds, do not fragment

watershed systems--i.e., exclude mixed ownership reaches or tributaries that would make the analysis ***"look good."***

- Conduct ESA, Section 7 assessments with journeyman-level fisheries biologists.
- Develop in-stream flow criteria for salmon watershed systems; provide adequate in-stream flows during critical low flow periods.
- ***Hold Resource Managers accountable for their decisions.*** Job performance criteria should include protection of salmon habitat. Replace managers insensitive to the protection of salmon habitat.
- In land management plan revisions, allocate salmon watersheds to salmon as the primary resource consideration.

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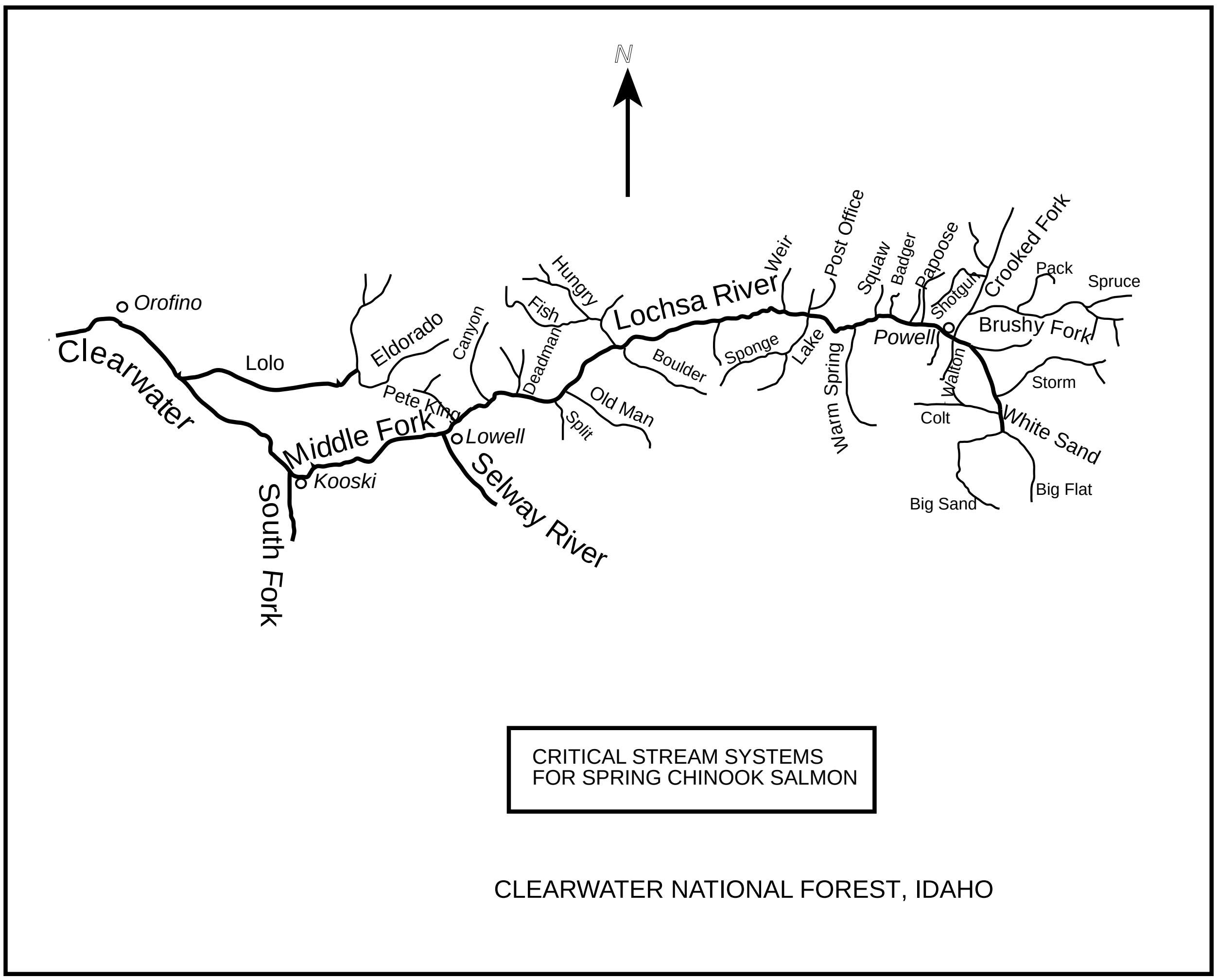

Figure B-1. Middle Fork Clearwater River and tributaries.

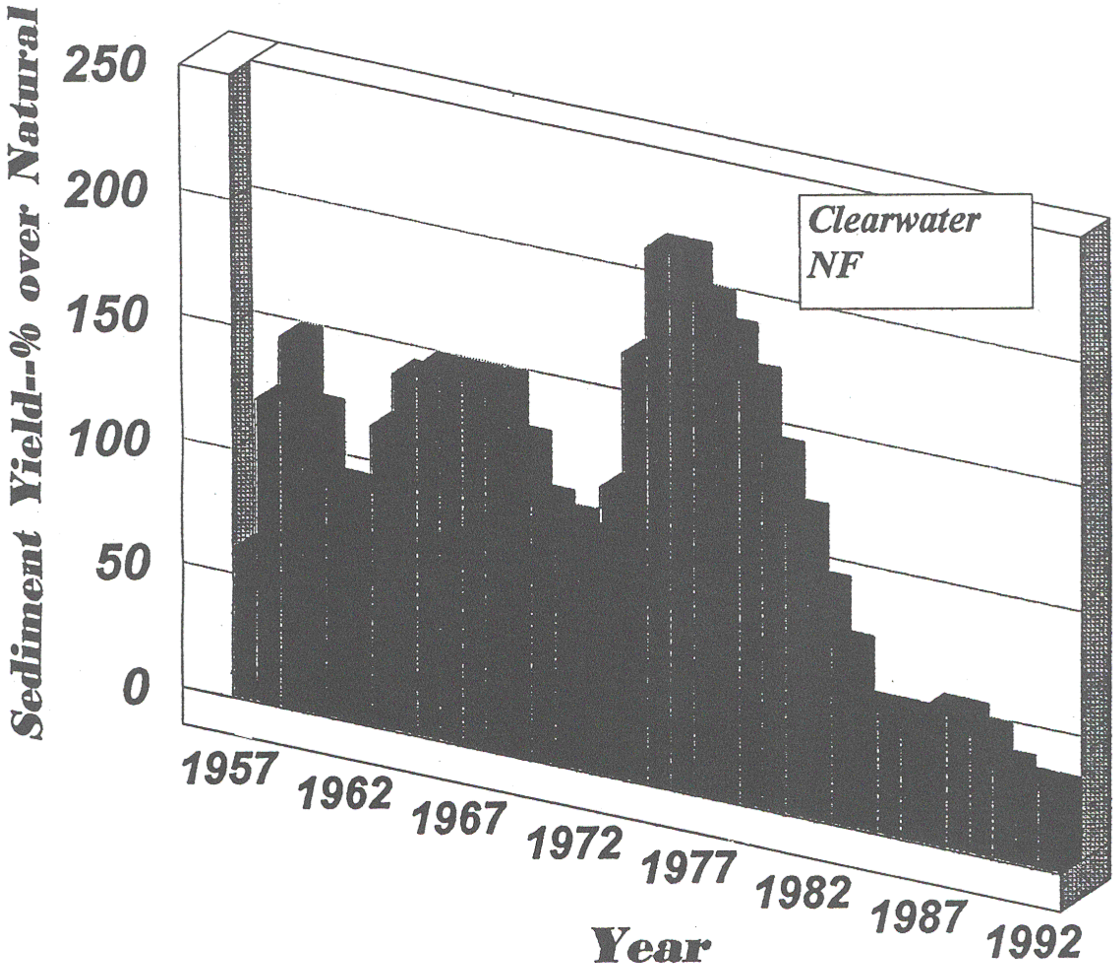

Figure B-2. Estimated sediment delivery from 1957 to 1992 in Lolo Creek on the Clearwater National Forest, Idaho (Clearwater National Forest, unpublished WATBAL runs).

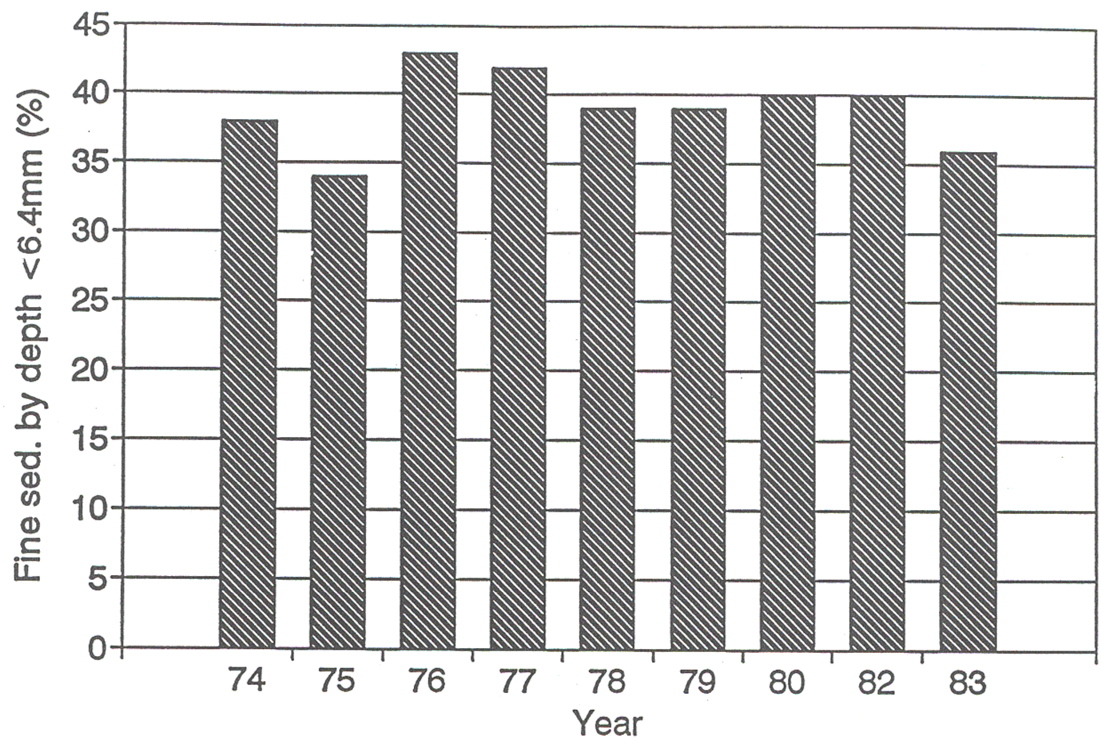

Figure B-3. Fine sediment by depth from coring in spawning substrate in Lolo Creek from 1974 to 1983 (Espinosa, unpublished data). Monitoring ceased in 1983.

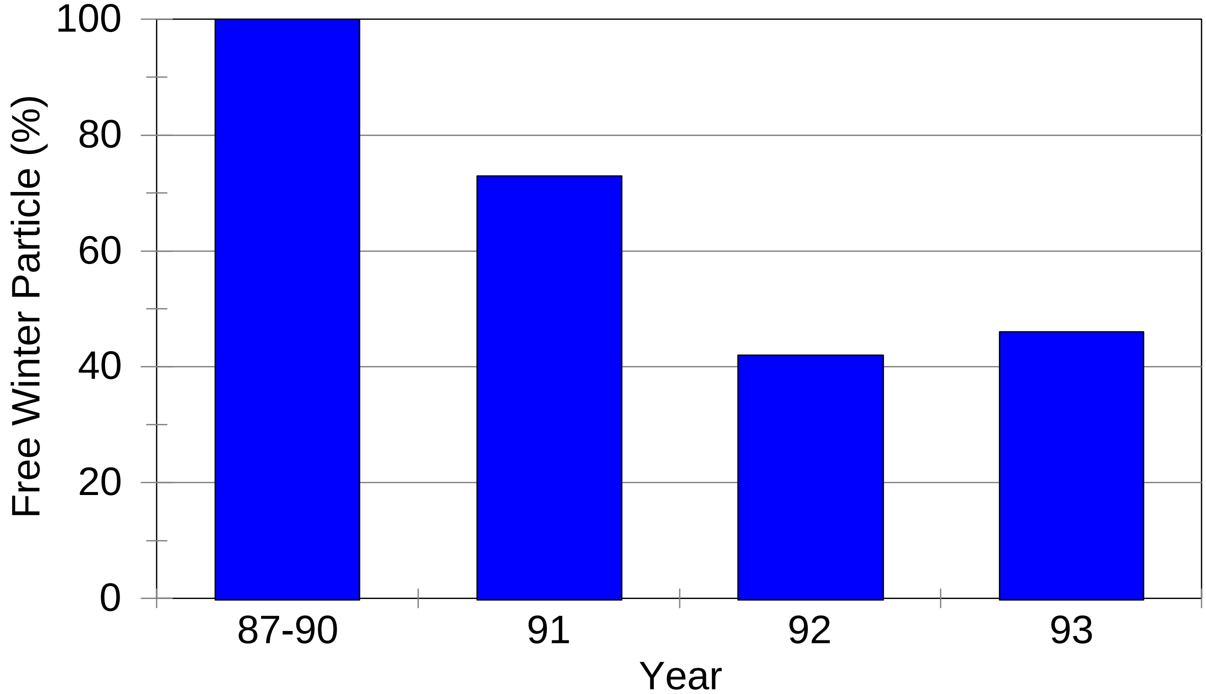

Figure B-4. Trend in "free winter particle" from 1987 to 1993 in Lolo Creek on the Clearwater National Forest (CNF) (Espinosa, unpublished data). Free winter particle is a measure of interstitial rearing space in winter habitat. Decreases in free winter particle also indicate in an increasing trend in cobble embeddedness (CE). Clean cobbles and boulders were added to the sample reach and the percent free winter particle (amount not embedded) was tracked over time. Data indicates that sedimentation continues to occur in winter rearing habitat under existing levels of sediment delivery. Annual differences in free winter particle were statistically significant ($p < 0.01$) (Espinosa, unpublished data).

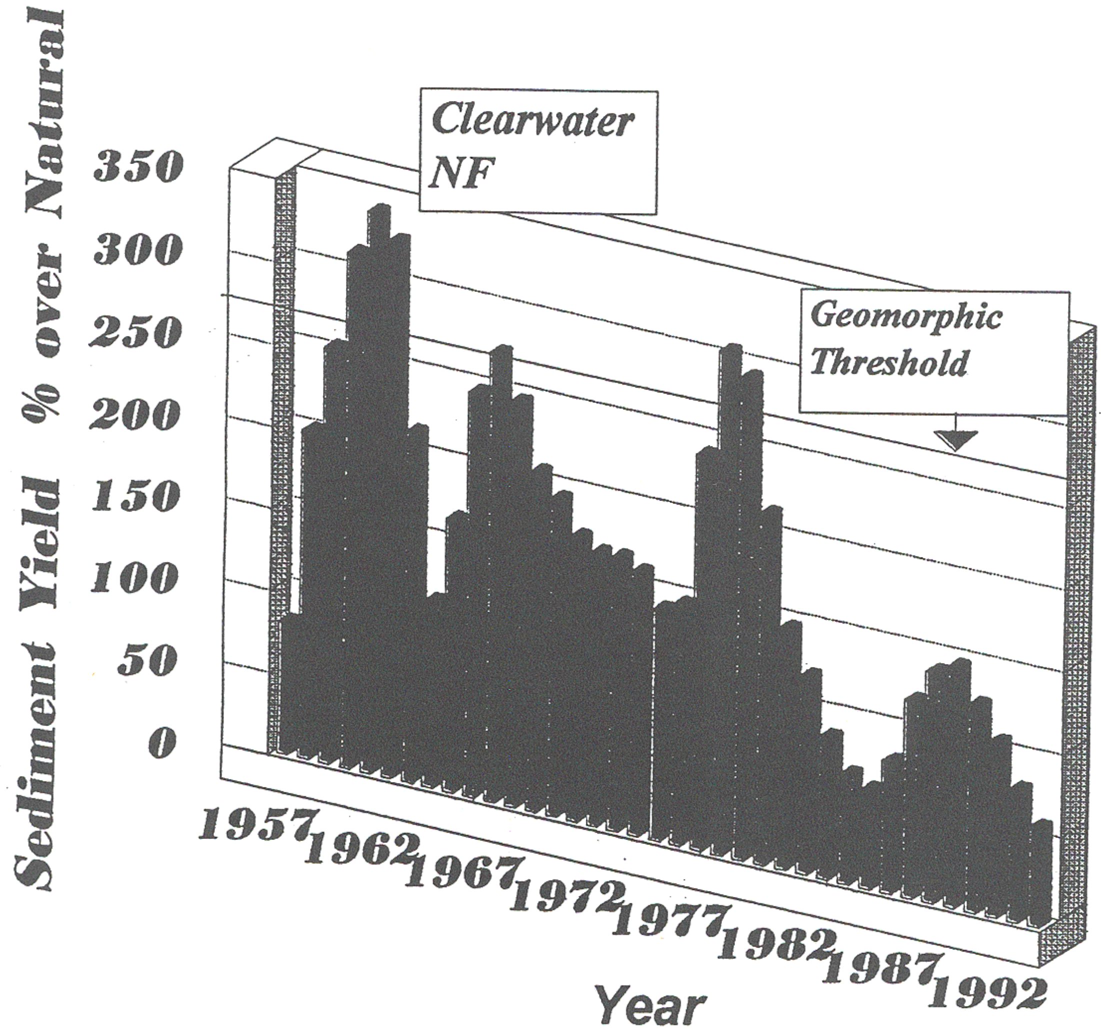

Figure B-5. Estimated sediment delivery from 1957 to 1992 in Eldorado Creek on the Clearwater National Forest, Idaho (Clearwater National Forest, unpublished WATBAL runs).

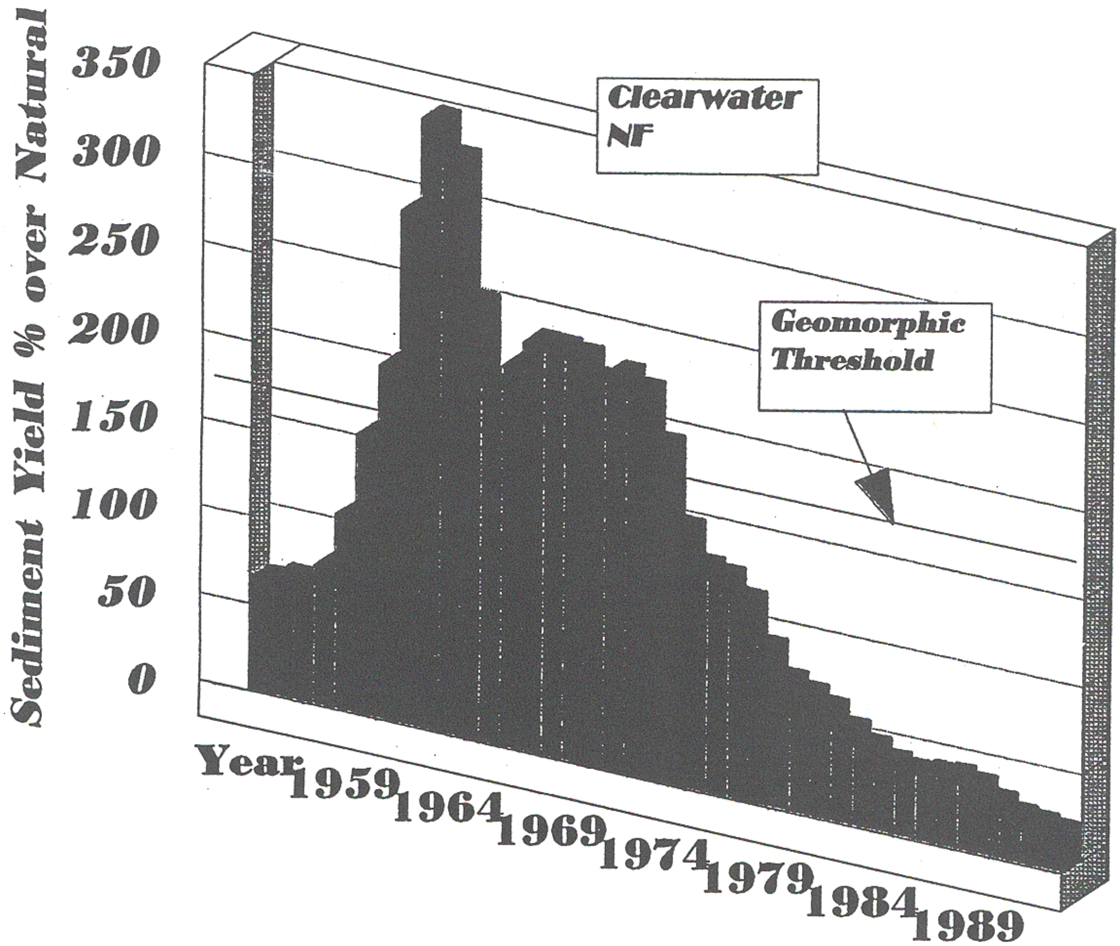

Figure B-6. Estimated sediment delivery from 1957 to 1992 in Pete King Creek on the Clearwater National Forest, Idaho (Clearwater National Forest, unpublished WATBAL runs).

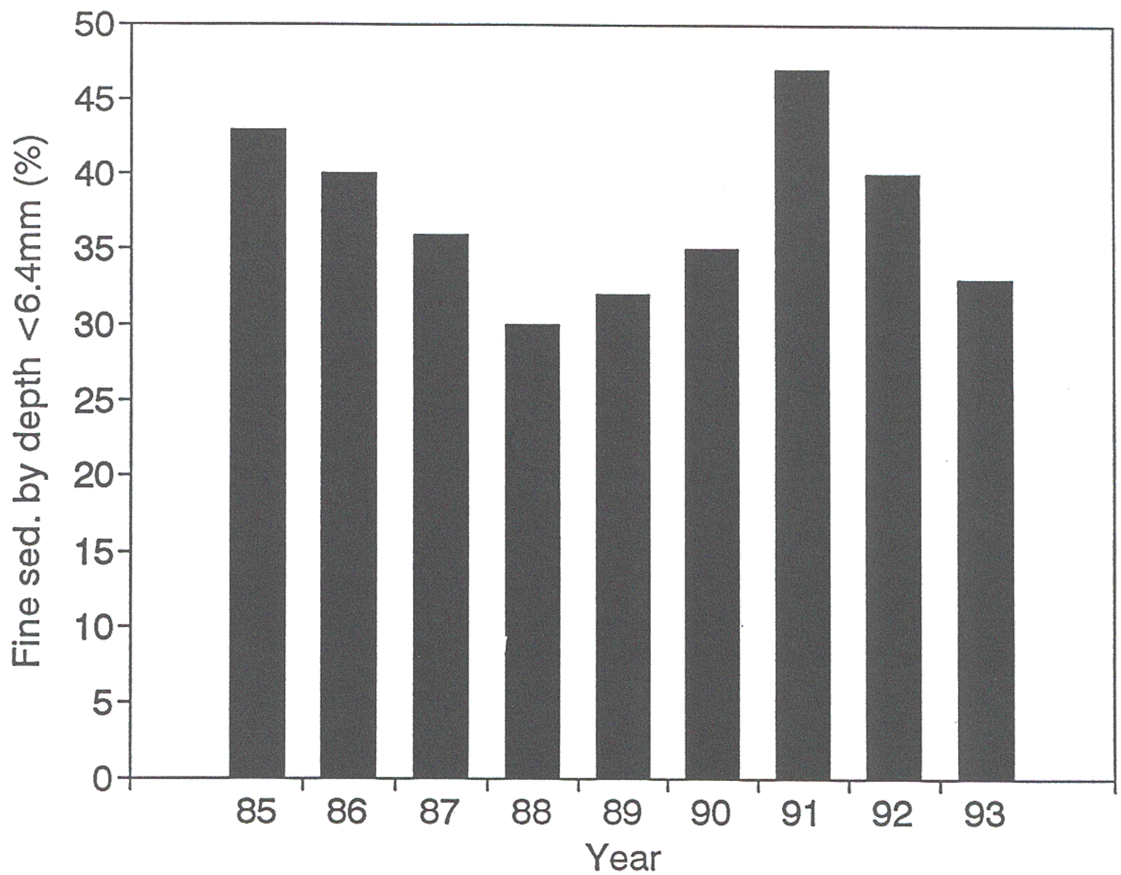

Figure B-7. Percent fine sediment by depth from coring in spawning habitat 1989-1993 in Pete King Creek on the Clearwater National Forest, Idaho. Sediment trapping and removal has occurred within the watershed since 1986.

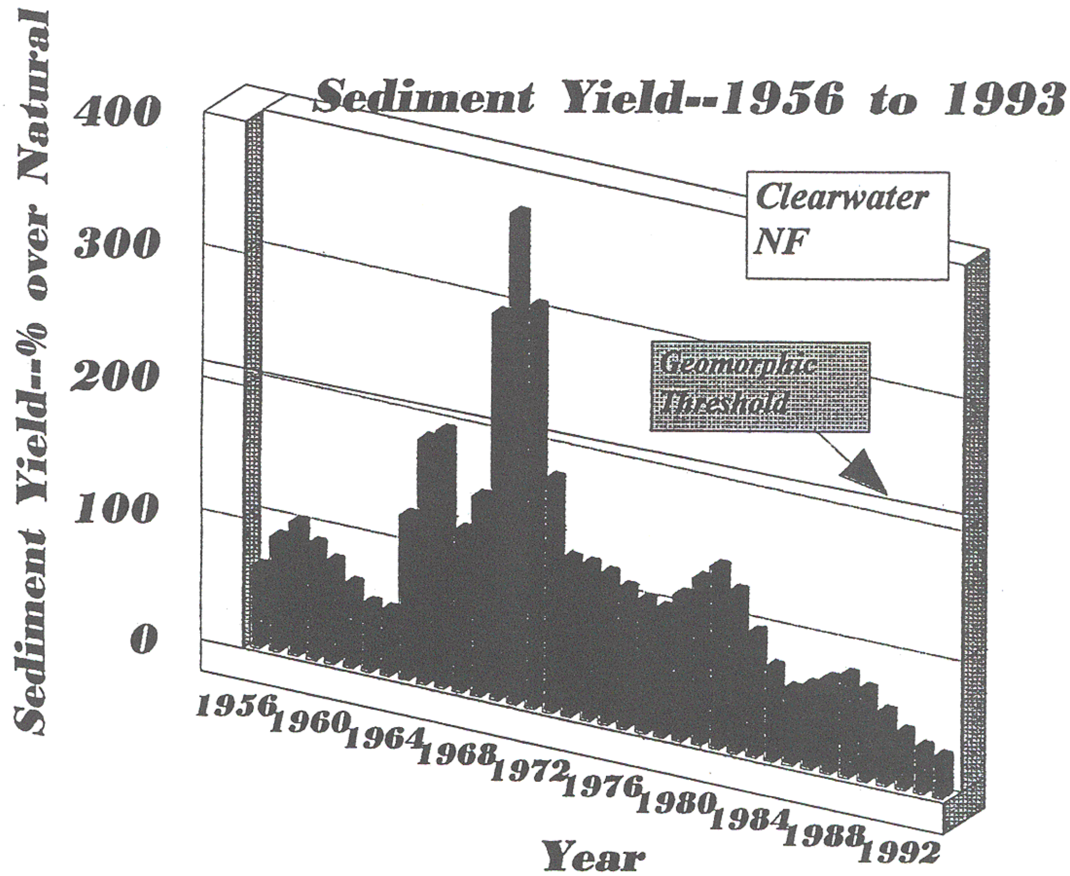

Figure B-8. Estimated sediment delivery from 1956 to 1993 in Squaw Creek on the Clearwater National Forest, Idaho (Clearwater National Forest, unpublished WATBAL runs).